Using insect biodiversity to measure the effectiveness of on-farm restoration plantings

by

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Declaration

This thesis contains no material which has been accepted for the award of any other degree or diploma in any tertiary institution, and to the best of my knowledge and belief, contains no material previously published or written by another person, except where due reference is made in the text of the thesis.

Vanessa Mann

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Annotation

This thesis is an uncorrected text as submitted for examination.

Abstract

Advances in farming technology, and the variety of modern agricultural practices, have the potential to reduce, maintain or improve biodiversity in an agricultural landscape. Environmentally sensitive farming systems are becoming more important on a local level, as climate change, declining biodiversity and habitat fragmentation impact the environment at a landscape scale.

Invertebrates are important components of an agricultural landscape, playing numerous roles including pest control, plant protection, pollination, and carbon cycling. They are also an important food source for many reptiles, birds, mammals and other insects, making them a key component of the food chain. Ants in particular are useful tools in biodiversity monitoring as they are abundant in both disturbed and intact habitats, and their many functional groups help to illustrate their community structure at a given point in time. For these reasons, they can be used to demonstrate the short and long term impacts of land management in various environments, including rehabilitated mine sites, fire affected regions, and agricultural landscapes.

Conducted on working farms, this study looked specifically at insect in the agricultural landscape, using 10 sheep pastures which have been restored with eucalypt plantings. Looking at species richness, relative abundance, and community structure, this study assessed the ant and beetle communities in these plantings and compares these to pasture control sites and nearby remnant woodland patch control sites. The influences of elevation, ground cover, soil clay, patch size, and age of planting were tested using regression analyses. It was found that leaf litter cover and weediness have a significant influence on invertebrate recolonisation of a restoration planting. Elevation was negatively correlated for all ant activity, whilst the age of the planting was positively correlated with ant abundance and species richness.

This study shows that ants can be useful monitoring tools in agricultural landscapes, and specifically useful when assessing the effectiveness of on-farm restoration plantings. It also provides a better understanding of the influence of environmental variables on a restoration planting, which in turn can help inform land management decisions.

Acknowledgments

First thanks must go to my supervisor Dr Peter McQuillan, without whose entomological expertise this research would not have even begun, and without whose continual enthusiasm this research would not have been finished. The many hours spent identifying my thousands of insects and helping me make sense of an overwhelming dataset is gratefully acknowledged.

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Chapter 1 Introduction

Modern agricultural practices have the potential to reduce, maintain or improve biodiversity in a productive agricultural landscape depending upon their application. Advances in farming technology and agricultural research enable landowners to adopt environmentally sensitive farming systems, which are becoming more important on a local level as climate change, declining biodiversity and habitat fragmentation increasingly degrade the environment at a landscape scale. These same advances in science also enable land managers and researchers to monitor changes in the landscape in real-time using a variety of biodiversity indicators and tools.

Terrestrial invertebrates are important components of an agricultural landscape because of their many ecological functions. Invertebrates act as natural biological control agents (Waterhouse & Sands 2001), recycle nutrients from plant litter (Fayle *et al* 2011), decompose the dung of native animals and livestock (Tyndale-Biscoe 1990), improve water infiltration rates thereby reducing erosion (Cerdà & Jergenson 2008), aerate and condition the soil (Cerdà & Jergenson 2008), and influence plant communities through pollination (Faegri & Van Der Pijl 1966), herbivory (Haines 1975) and seed dispersal (Howe & Smallwood 1982; Hughes & Westoby 1992). They occupy many niches, from narrowly specialised roles as parasites on a single host to generalised scavengers on dead root material. Invertebrates are therefore a versatile tool to assess a variety of impacts of land management techniques or land restoration.

Ants in particular can be useful tools in biodiversity monitoring in a variety of ecosystems, especially in Australia where ant species richness is very high. Ant communities have been shown to respond significantly when their environment is altered therefore making excellent indicators of change. For example, ants have been used successfully to monitor minesite rehabilitation (Andersen 1993), the effects of pesticides in agricultural landscapes (Batàry 2012), recovery from bushfire (Andrew *et al* 2000; York 2000) and progress in land conservation measures (Majer 1985). The response of ant communities to land management and land use decisions can be sampled easily and cost effectively, and does not necessarily require a high degree of specialised entomologist input (Andersen *et al* 2002). Literature and internet resources are now readily available to assist people without entomological expertise identify most Australian ants to at least a genus level. Reviews of species level identification, including keys, have been recently published for several important, but previously intractable, genera such as *Iridomyrmex* and *Monomorium*. Ants therefore are ideal tools to use as indicators of biodiversity and ecosystem health. Using ants to monitoring ecological changes in a restored landscape over time, identifying environmental factors which influence these changes, can help to inform future land management decisions.

1.1 Overview of the Study

Renewed interest in landscape health following evidence of large scale dieback in farm trees in the Midlands and elsewhere (Close & Davidson 2004) has led to attempts to restore local trees in plantings. Over the last two decades, Greening Australia has worked with land owners to establish demonstration-scale plantings across a number of catchments in eastern Tasmania. One motivation for this is to test whether biodiversity more generally is advantaged by this process, with a view to harnessing the ecological benefits to the long term advantage of farmers; for example, through less erosion or less need for pesticides.

This project aims to determine whether a set of pitfall traps set in different restoration plantings can demonstrate any changes in the invertebrate communities, specifically ants and beetles, at those sites.

1.2 Research Aims

The null hypothesis of this study is that there will be no difference in the relative abundance, species richness, and community structure of ants or beetles between restoration plantings and in nearby pastures. The aims of the study therefore are:

- To determine whether there is a significant change in the ecological measures of species richness, relative abundance, and the ant and beetle community structure in a group of restoration plantings, compared with pasture controls.

- To consider the ant community in a restoration planting and determine whether it is more similar to that in the source paddock or more similar to a nearby remnant patch.

- To determine whether any environmental factors, alone or in combination, affect the recolonisation of a restoration planting by invertebrates. Environmental factors for consideration include soil clay content, the size of the planting, elevation, proximity to a remnant woodland patch, and ground cover and weediness.

1.3 Definitions of "plantation" and "planting" as used in this thesis

It is important to define two potentially interchangeable words used in this thesis – "plantation" and "planting."

The Australian Forestry Standard (2007, p17) defines plantations as ''*Stands of trees with native or exotic species, created by the regular placement of cuttings, seedlings or seed selected for their wood-producing properties and managed intensively for the purposes of future timber harvesting*."

In this thesis, areas which have been revegetated with *Eucalyptus* spp. and other native trees and shrubs, for the purposes of landscape restoration, are referred to as plantings. Areas which have been planted with monoculture *Eucalyptus* spp, or other monoculture tree species, for the purposes of commercial wood crops, are referred to as plantations.

The key difference between a plantation and a planting is the commercial purpose of the plantation. In addition, plantations are often subject to high levels of agrochemicals for fertilisation or pest control, and therefore may display different characteristics to a planting not exposed to these chemicals. Plantings on a working farm on the other hand, may have undergone some weed control by the farmer, and may be exposed to agrochemicals used at their perimeters.

Chapter 2 Background

2.1 Landscape Restoration

At the time of European settlement around 70% of Australia was covered by woody vegetation. In the 225 years since then, an area of 92.5m ha has been cleared. Analysis of satellite data has shown that about half of this clearing was to facilitate grazing by farmed animals (Barson *et al* 2000). In the Midlands region of Tasmania, native vegetation was reduced to 16.9% of its original area by 1985 due to clearing for agricultural development (Fensham & Kirkpatrick 1989). This extraordinary rate of land clearing continues today, with the estimated rate of decrease in woody vegetation due to clearing for agriculture and grazing being 292,030 ha per year across Australia (Barson *et al* 2000).

Tree clearing results in changes in hydrology patterns, increasing water runoff and soil erosion, accelerating processes such as salinity and water logging, as well as a causing a loss of natural habitat for native animals. Native tree decline caused not by purposeful land clearing, but by poor land management practices and climate change is a widely observed trend in rural areas across Australia, and is particularly severe in the Midlands region of Tasmania (Close & Davidson 2004). Species richness in Australian agricultural environments is nearly always lower than in naturally vegetated habitats (Lobry de Bruyn 1999), so this strongly suggests that further tree clearing will result in further loss of species richness. Habitat loss associated with land use changes following development is one cause of the biodiversity decline which has occurred since European settlement in 1788 (Swanson 1995).

More enlightened land management decisions have the potential to reduce the rate of tree clearing and rural tree decline, or even reverse it. Revegetation and habitat restoration programs are becoming commonplace across Australia, and are frequently targeted to agricultural landscapes. On a larger scale, landscape restoration is the process of improving degraded and destroyed landscapes or ecosystems. This can be achieved through various methods including enhancing soil fertility, modifying plant communities, and reversing the effects of mining, agriculture or logging through planting new vegetation. Restoration planting involves a number of processes – establishment, succession, and dispersal – and the rate and quality of restoration are affected by environmental variables, both above ground and underground (del Moral *et al* 2007).

Whether the restoration involves planting a shelterbelt of trees along paddock edges, or rehabilitating entire pasture sites to woodland, there may be a range of ecological benefits (Tongway & Ludwig 2011):

- Providing corridors for movement of native animals across the landscape
- Providing shade and shelter for farmed animals
- Reducing soil erosion and water runoff
- Improve soil properties through producing leaf litter

An NGO, Greening Australia, facilitates restoration projects including the very large scale Gondwana Link in Western Australia, where certain areas have been identified for protection and restoration to become part of a 1000 km contiguous stretch of natural bushland (Greening Australia 2013). This will help re-connect what has become a highly fragmented landscape. Greening Australia also facilitates many smaller projects on private land across the country including many in the Midlands region of Tasmania, some of which have become study sites for this research. Whatever the scale, revegetation and landscape restoration programs can not only contribute to preservation of habitat for biodiversity, but can also play a role in carbon sequestration for climate change mitigation.

Site-based monitoring of either specific species or communities of species can help us understand the effects of landscape restoration, and help assess the management actions that can slow or reverse rates of biodiversity decline. The important first step is to collect baseline data and establish ongoing monitoring programs that can help to inform management decisions. This study may become the foundation of one such program.

2.2 Habitat fragmentation and the agricultural matrix

Agricultural landscapes often consist of a diverse mosaic of vegetation types, including sown pastures of exotic grasses, fields of native grasses, agricultural crops, heterogeneous hedgerows, mixed species shelterbelts, and remnant native woodlands. These remnant woodlands are often the only remaining examples of the original habitat type, making their conservation an important priority (Fensham & Kirkpatrick 1989).

Conservation of non-agricultural components of a matrix may be affected by a number of factors, including the presence of grazing stock, the use of pesticides and fertilisers in parts of the matrix, weed invasion, and even by the quality of the farmland within the matrix, which serves as habitat corridors linking fragments of suitable habitat for native flora and fauna.

Over-grazing by hoofed mammals can have a huge impact on the integrity of remnant woodland and other native vegetation types. It can prevent regeneration of native vegetation, diminish the leaf litter volume at ground level; and through soil compaction, grazing changes soil moisture content, increases runoff, and promotes erosion (Bromham *et al* 1999). This is in contrast to the benign impact of the softer Australian macropod foot, which has a larger surface area relative to the weight of the animal when compared with cattle and sheep.

Establishing or maintaining a matrix of diverse vegetation types in an agricultural landscape is thought to be highly beneficial to avian fauna (Fischer *et al* 2005). Maintaining tree cover on just 10% of an agricultural landscape can have a considerable benefit to avian fauna in that habitat (Bennett & Ford 1997). However, the establishment of an agricultural matrix may make only a “modest contribution” to the ant species richness (House *et al* 2012). Or, conversely, the effects of habitat fragmentation may be considerably weaker on ant communities than on other animals in the landscape.

Within a remnant patch, changes in habitat can have a significant effect on invertebrates in that habitat (Debuse *et al* 2007). Changes to the within-patch characteristics of a remnant patch are more influential on species composition than is the nature of the wider landscape matrix (Debuse *et al* 2007). It would follow then, that changes within a restoration planting could have an impact on its resident invertebrates – potentially positive or negative, depending on the nature of those changes.

Hard edges between suitable habitat and farming land may have an impact on ant assemblages, causing a significant difference between ant communities inhabiting the area near a sharp edge and those species in the centre of a field of fallow land (Dauber & Wolters

2004). Hard edges created by mowing, fences, or natural boundaries such as rivers, may deter movement of ground-dwelling invertebrates from one habitat type to the next.

One theory suggests that the edge-area ratio of a patch affects the abundance of ants and beetles rather than patch size being the main indicator of relative abundance within that patch. A relatively greater number of ground-dwelling beetles was found to inhabit grassland patches with lower edge length by Golden & Crist (2003). The key is to have a suitable area of native invertebrate habitat conserved in the landscape, whether that be a remnant patch of native grasses, a stand of paddock trees, a mixed species shelterbelt, or a restoration planting. Within a farming matrix, invertebrate species richness in a remnant woodland patch can be significantly higher than in other vegetation types (House *et al* 2012); as it is in a stand of paddock trees compared to surrounding grazing pasture (Oliver *et al* 2006). These studies emphasises the need to identify and conserve existing suitable habitat within an agricultural landscape. Appropriate management of the unfarmed parts of a landscape is an important principle for invertebrate conservation (New 2005).

Many farming environments however are now devoid of large patches of native woodland remnants. In these environments, revegetation of parts of the farming landscape may facilitate positive changes in invertebrate biodiversity. This study looks specifically at whether the practice of restoring paddocks used for stock grazing to a native woodland condition would result in an increase in insect biodiversity.

2.3 Existing monitoring practices

Restoration is a unique form of disturbance in a landscape, because it alters the environment in ways which in turn alter species composition (del Moral *et al* 2007). Current monitoring practices in restoration ecology rarely consider the presence of insects when assessing changes in species composition at a site. When restoration programs take place in agricultural landscapes, woodland environments, and even forestry coupes, it is usually the terrestrial vertebrates (Lindenmayer *et al* 2000), the avian fauna (Grey *et al* 1997; Fischer *et al* 2005; Benton 2007; Firbank *et al* 2008, Tongway & Ludwig 2011), overall plant biomass or species composition (Lindenmayer *et al* 2000; Firbank *et al* 2008; Batáry *et al* 2012) which becomes the focus. In Germany, invertebrates such as carabid beetles and spiders have been used to assess the impact of nitrogen fertilisers and pesticides in a cropping landscape (Batáry *et al* 2012). However, invertebrates are rarely used as biological indicators of restoration success or of the health of an agricultural landscape and ants in particular are under-represented in monitoring practices in agriculture; indeed, soil invertebrates have been called “a largely forgotten component of biodiversity” when it comes to pasturelands (Tongway & Ludwig 2011). A recent review of faunal responses to revegetation (Munro *et al* 2007) looked at 27 studies, in which only 4 considered invertebrate fauna, and only 2 of these focused exclusively on invertebrate fauna. Only one study could be found in which the researchers monitored the change in ant communities on land previously used for agriculture and now revegetated with eucalypts. Schnell *et al* (2003) recorded a significant change in the ant species richness in a monoculture *Eucalyptus punctata* plantation established on pasture, when compared with results from sampling the same plantation 6 years prior. A Tasmanian study of ground dwelling invertebrates in forestry plantations sampled 17 monoculture *Eucalyptus nitens* plantations, 26 monoculture *Pinus radiata* plantations, and 3 monoculture *Pseudotsuga menziesii* (Douglas fir) plantations (Bonham *et al* 2002). Looking specifically at land snails, carabid beetles, millipedes and a threatened velvet worm, it was found that the invertebrate communities in the pine plantations resembled that of surrounding native forest more closely than did the *E. nitens* or Douglas fir plantations. However, no ants were included in this study.

Common practices in agriculture have the ability to cause a decline in ant biodiversity. The use of fertiliser and pesticides can affect populations directly or indirectly by poisoning their food supply, and tilling the soil can damage nests and cause a reduction in leaf litter and organic matter (Lobry de Bruyn 1999). Typically, ants are under-recognised and undervalued by landowners in farming landscapes. For example, according to a group of 28 Canadian farmers, ants are not even considered when thinking about indicators of soil health (Ronig *et al* 1995). Unprompted, these farmers nominated a number of direct indicators including soil texture, soil colour, ease of tillage, presence of earthworms, and calcium and magnesium content, as measures of soil health. They were quick to accept a soil test, yet unsure of what constitutes a good level of certain minerals. However, these same farmers widely considered indirect indicators such as crop rotation, crop yield, weediness, chemical use, and even healthy cows as measures of soil health. Not one farmer considered the value of ants in the landscape and yet ants can be readily used as indicators of soil health in an agricultural landscape, with the potential to provide an early warning of problems such as contamination, or negative trends in soil quality (Lobry de Bruyn 1999).

Whilst the suitability of invertebrates as biodiversity indicators in agriculture has been recognised and utilised by only a minority of scientists and few landowners, their use in the assessment of disused minesite rehabilitation has become widespread. Ants in particular are highly sensitive to environmental toxins, keeping a much greater distance from contaminated land than mammals do in the same environment (Andersen *et al* 2002). Whilst much research has been conducted into the utility of various monitoring tools, ants have been shown to be effective indicators for the success of minesite restoration projects (Andersen 1993; Bisevac & Majer 1999; Andersen & Majer 2004; Moir *et al* 2005; Williams *et al* 2012) especially when used in conjunction with other monitoring tools (Underwood & Fisher 2006). If they can be useful in monitoring changes stemming from minesite rehabilitation, then ants and other invertebrates must have similar potential to monitor changes in an agricultural landscape.

2.4 Economic value of insect conservation

There is an economic benefit to be gained from insect conservation in agricultural landscapes. Insects deliver multiple ecological benefits including crop pollination, pest control, decomposition of animal waste, and nitrogen cycling. Calculations estimate that the economic value of having wild, native insects playing these critical roles in farming environments is worth around $60 billion per year to the US economy alone (Losey & Vaughn 2006). This doesn't include the value of insects bred for large-scale biological controls, nor domesticated bee-keeping exercises, nor any products produced commercially by insects such as silk, shellac, honey, or wax. It follows then, that farmers in Australia would also experience an economic benefit if pollination, pest control, dung decomposition and nitrogen cycling functions were carried out, at least in part, by wild, native insects. The challenge for landowners will be in making their farms more attractive to insects when the very nature of agriculture tends to deplete biodiversity at a local scale (Benton *et al* 2003; Nicholls *et al* 2001).

2.5 Ant functional groups

Invertebrates are arguably the backbone of life on earth. An estimated 990,000 species of invertebrates have been described, as opposed to approximately 42,580 species of vertebrates (Wilson 1987). If invertebrates were to experience a mass extinction, then most of the fishes, amphibians, birds and mammals would disappear as a consequence due to the failure of symbiotic relationships and gaps in the food chain.

Among the most distinctive of terrestrial invertebrates are the ants, representing a single family of social insects with approximately 8,800 species described. Australia is particularly rich in ant taxa, supporting an estimated 3,000 species, of which 140 are found in Tasmania.

Ants play a diverse range of roles within ecosystems, including seed harvesting and dispersal, predation of other ants and insects such as termites, fungi regeneration, and assisting with the decomposition of organic matter. Following Andersen (1995), Australian species can be categorised into the following functional groups:

- Dominant Dolichoderinae – including *Anonychomyrma* spp. and *Iridomyrmex* spp.
- Subordinate Camponotini – including *Camponotus* spp. and *Polyrhachis* spp.
- Hot Climate Specialists – including *Melophorus* spp. and *Meranoplus* spp.
- Cold Climate Specialists – including *Notoncus* spp. and *Prolasius* spp.
- Tropical Climate Specialists – including *Oecophylla* spp and *Polomyrma* spp.
- Cryptic Species – including *Amblyopone* spp. and *Solenopsis* spp.
- Opportunists – including *Rhytidoponera* spp. and *Tapinoma* spp.
- Generalised Myrmicinae – including *Pheidole* spp. and *Crematogaster* spp.
- Specialist predators – including *Cerapachys* spp and *Epopostruma* spp.

Due to their range of functions, ants can be used as indicators through not only relative abundance and species richness measures, but also by analysing the community structure by considering the above functional groups. Changes in a landscape or study area may be measured by monitoring changes in the ant community structure over time, and this has been proven to be an effective monitoring tool in *Eucalyptus* plantations (Schnell *et al* 2003). A healthy ant community would generally be one with a range of functional groups present, not one dominated by a single group.

Alternatively, the monitoring of just one functional group can provide indications of whether land management decisions are having an impact at a smaller scale. For example, a large amount of research has been conducted into the contribution of ants in coffee plantations in South America and Africa. Ants have been shown to act as valuable natural biological control agents in Mexican coffee plantations (Vandermeer *et al* 2002). However, as a result of coffee farming intensification, ants are experiencing widespread habitat loss in Latin America (Perfecto *et al* 1996; Perfecto & Snelling 1995; Philpot and Foster 2005; Philpott *et al* 2008). By focussing on this one group of Specialist Predators, researchers can evaluate land management decisions to assist in the preservation of suitable nesting habitat within coffee plantations, which in turn may improve productivity levels.

This trend is reflected in Australian farming landscapes where land use intensification has resulted in large areas of habitat loss for native invertebrates.

Chapter 3 Materials and Methods

3.1 The Study Area

This study was conducted at 28 rural locations in south east Tasmania between the latitudes 42.2422°S and 42.5724°S, and the longitudes 146.7723°E and 147.6139°E (see Figures 1 and 2 for detailed maps). This area spans the Upper Derwent Valley including the towns of Bothwell, Melton Mowbray, and Hamilton, and the adjacent Midlands region covering the town of Oatlands and surrounds.

The climate is cool temperate, with mean maximum January temperatures of 22.5°C and 23.9°C recorded at local weather stations in Bothwell and Melton Mowbray respectively, and a mean maximum January temperature of 21.7°C in Oatlands. Annual mean rainfall for the region ranges from 444.8mm in Melton Mowbray to 548.7mm in Oatlands (Australian Bureau of Meteorology data accessed 16 June 2013, http://www.bom.gov.au/climate/data/). Climate data is shown in Table 1.

Table 1. Average temperatures and rainfall for the study area.

	Bothwell	**Melton Mowbray**	**Oatlands**
January mean minimum temp (°C)	7.5	10.3	8.7
January mean maximum temp (°C)	22.5	23.9	21.7
July mean minimum temp (°C)	-0.2	1.9	1.1
July mean maximum temp (°C)	10.6	11.3	9.4
Average annual rainfall (mm)	536.7	444.8	548.7

The major land use is agriculture based largely on livestock grazing on perennial pastures. In recent decades, agricultural enterprises have diversified and annual cropping, viticulture and monoculture plantations have been increasingly pursued. Long term decline in some landscape conservation values, manifest in tree dieback, erosion and pest outbreaks, has seen rising interest in revegetation and restoration projects.

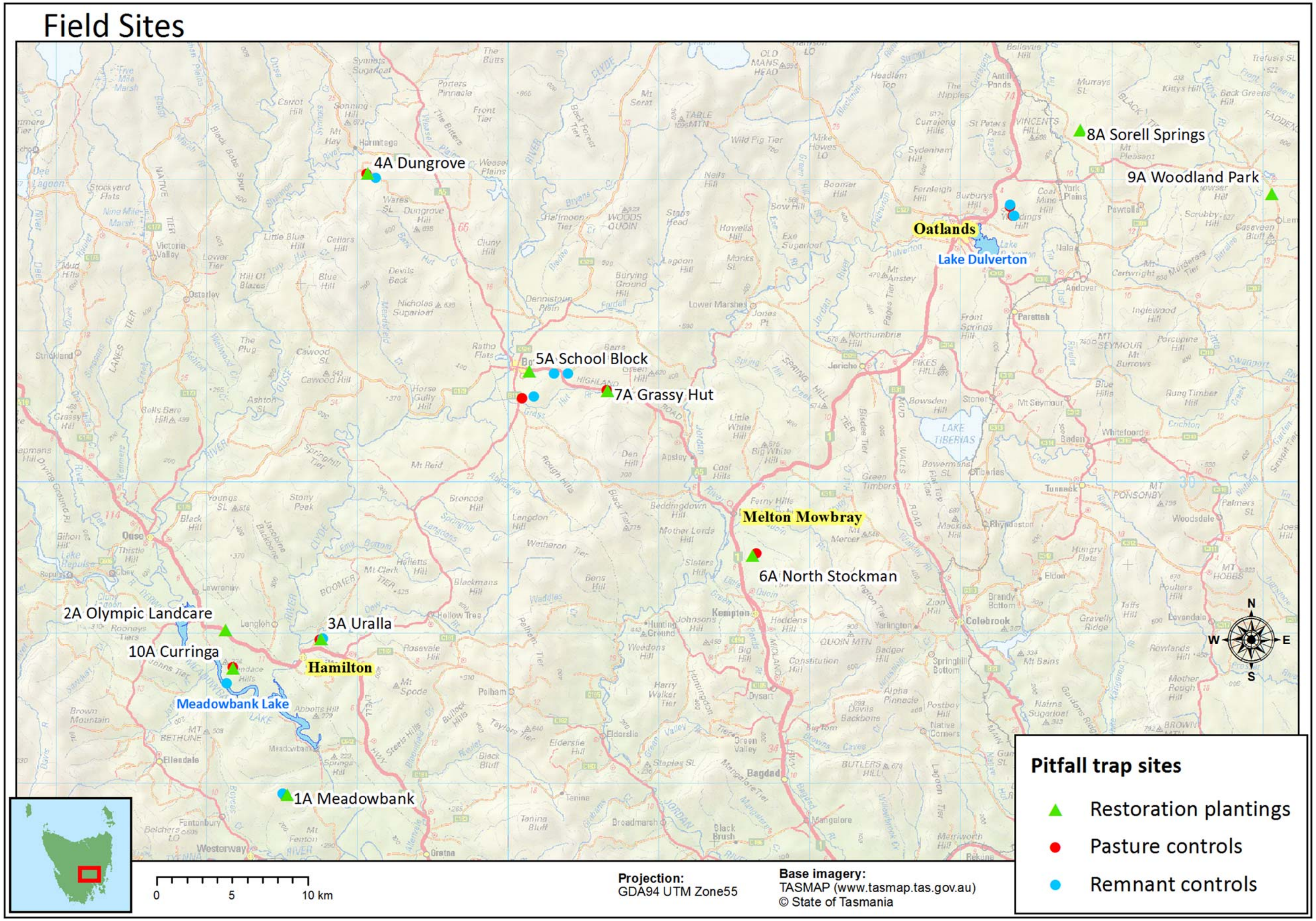


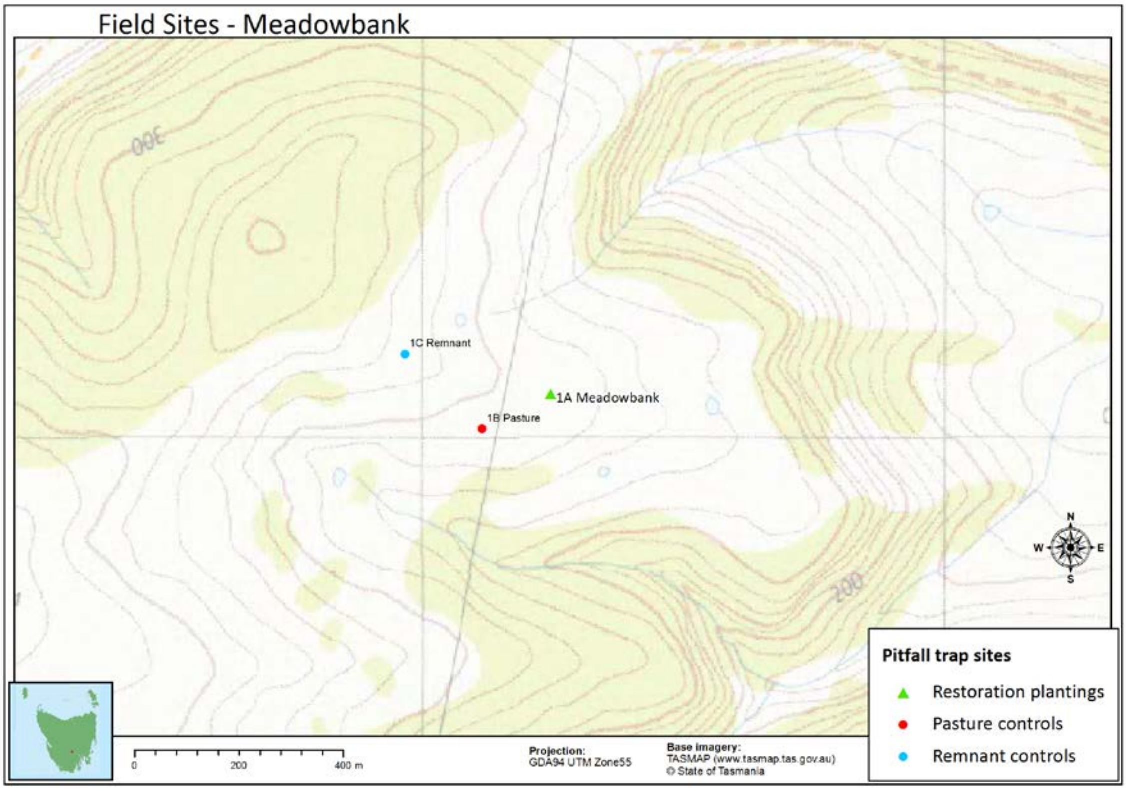
Figure 1. Location of the 10 study sites, all named in the map, and the nearby controls, shown with coloured markers.

3.2 Research design

Ten restoration plantings were chosen as sampling locations, with the support of Greening Australia, who initiated, cultivated, and provide ongoing management support for all plantings. To avoid selection bias, none of the sites had previously been the subject of invertebrate fauna surveys. They were selected to show a range of ages and conditions. All plantings were former sheep paddocks on private tenure which had been offered by the landowners as sites for rehabilitation. The rehabilitation process for every site has followed a consistent process which includes soil preparation, weed eradication, the planting of seedlings in rows, and erecting fencing to exclude stock from the site (Davidson 2013). Seven of the plantings have *Eucalyptus pauciflora* as the dominant species, with smaller numbers of *E. tenuiramis* and *E. rubida* included in these plantings, all of local provenance. A further two plantings were direct seeded by the landowners, straying from the practice of establishing a plantings using seedlings. These two sites display a mixed species regeneration containing *Acacia melanoxylon, Dodonaea viscosa, E. globulus, E. delegatensis,* and *E. leucoxylon*. The latter eucalypt is not native to Tasmania. The remaining site, in a peri-urban area, was planted with seedlings of *E. amygdalina, E. viminalis,* and *E. pauciflora*. The plantings display a variety of understory and ground cover, ranging from bare soil and exotic grasses, to a mixture of leaf litter, native grasses, lichen and moss.

The sites are all located in south eastern Tasmania, with four in the Upper Derwent Valley in the localities of Lake Meadowbank, Hermitage, and Hamilton; four in and around Bothwell, and two near Oatlands in the Upper Midlands region. Nine of the plantings were former sheep pastures on working farms, and one restoration planting is in a peri-urban area. These plantings were specifically selected as they represent a range in ages from 2 years to 25 years, and a range of altitudes from 113m to 552m. Their distance to the nearest remnant patch varies from being adjacent, to approximately 3100m away.

Photos of all sites are shown in Figure 4, with detailed maps of the study locations showing topographic features in Figure 2.



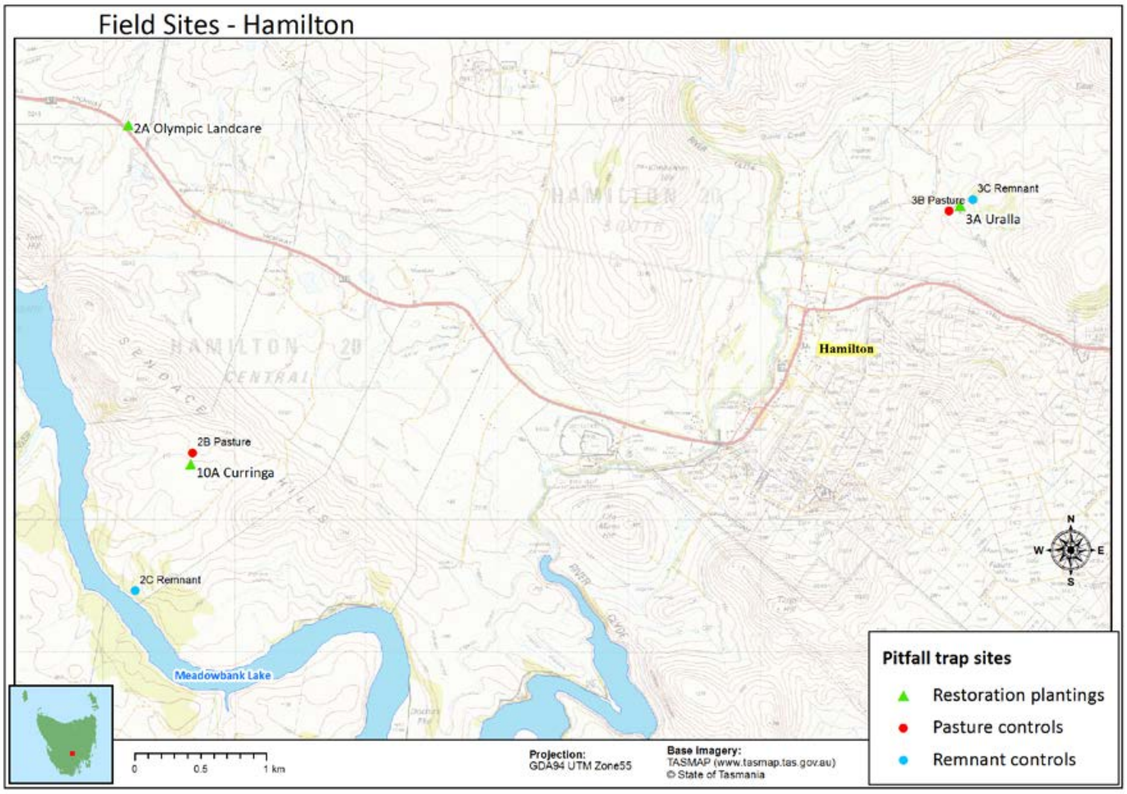
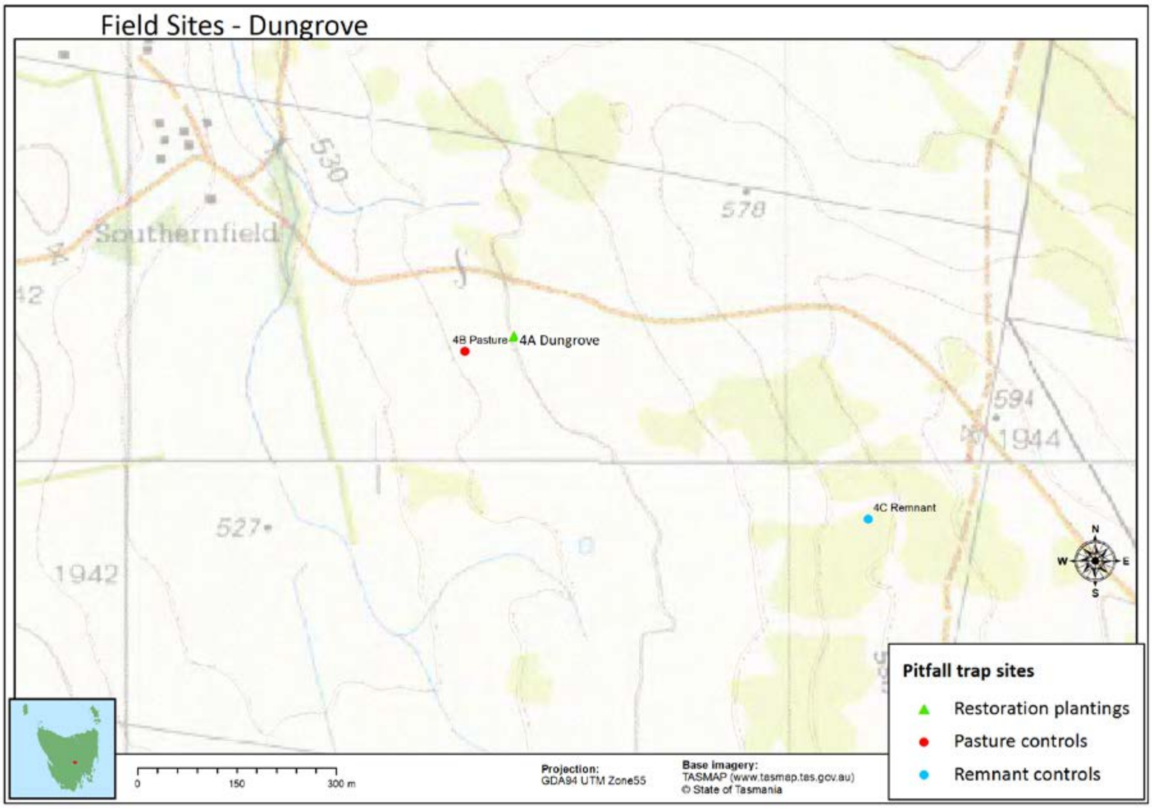
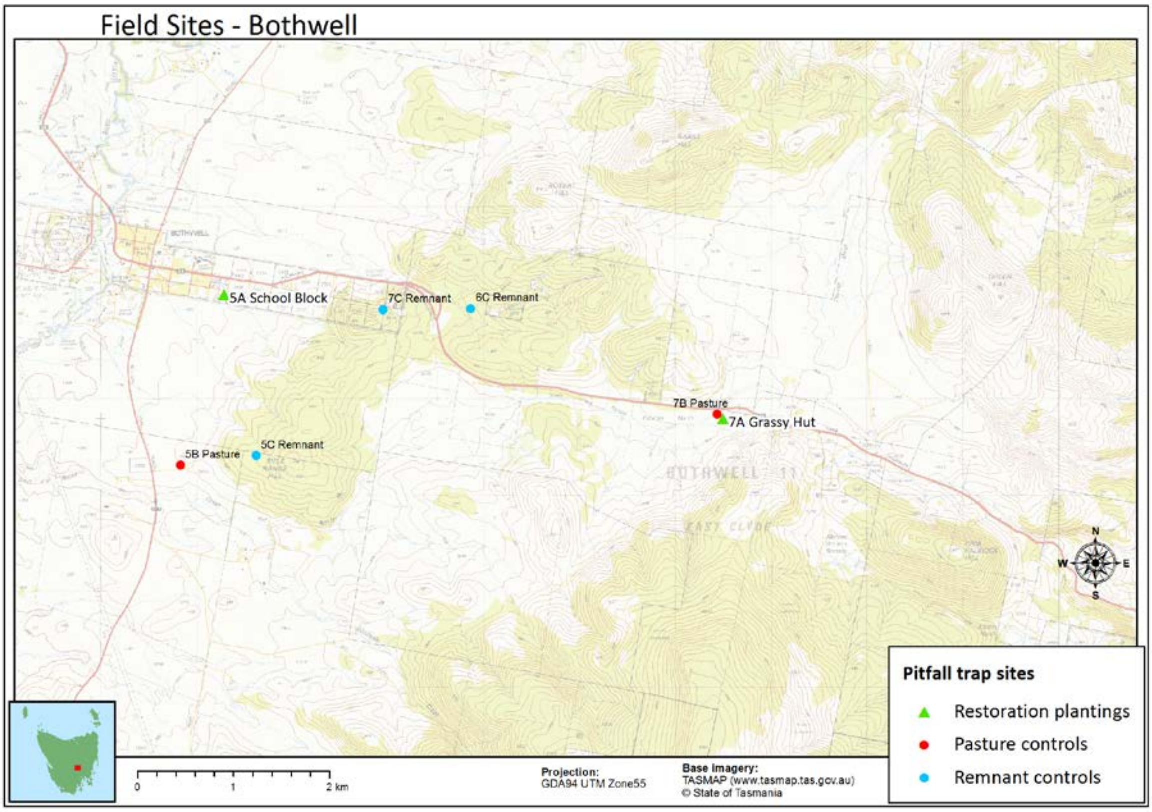


Figure 2. Detailed maps of the study locations showing topographic features.





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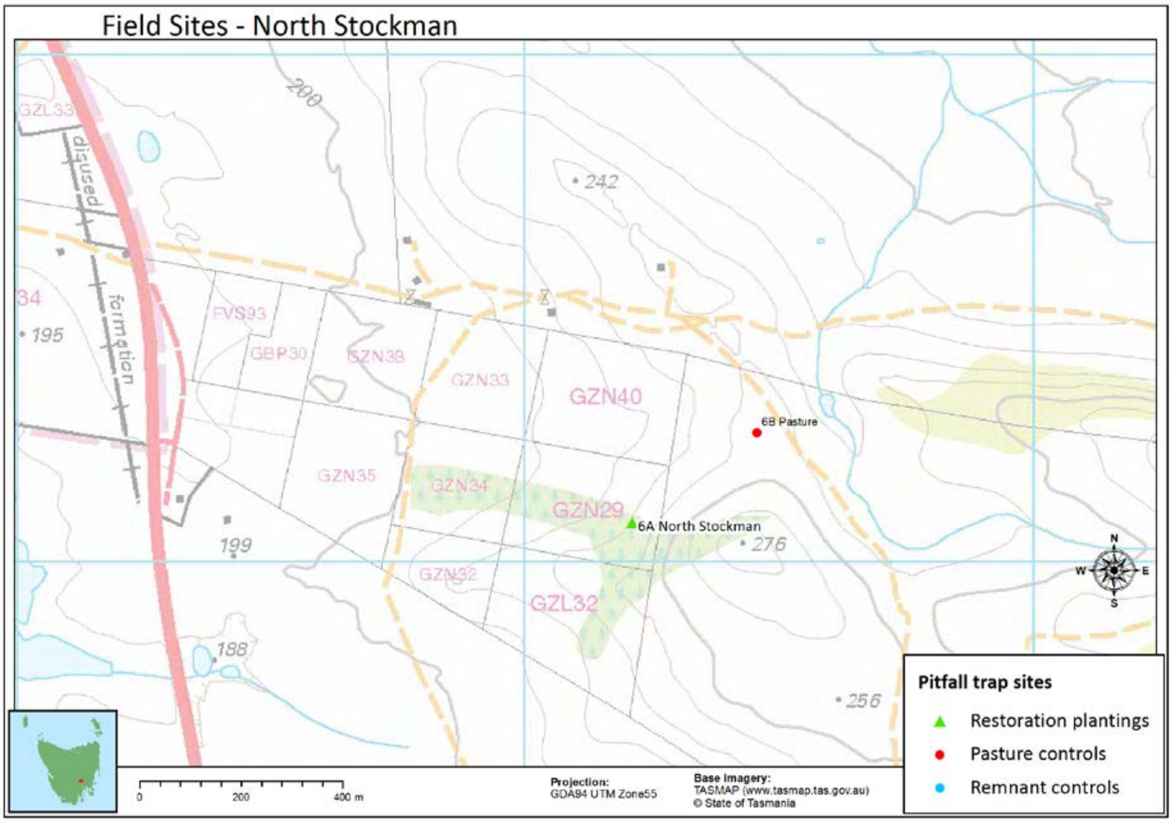

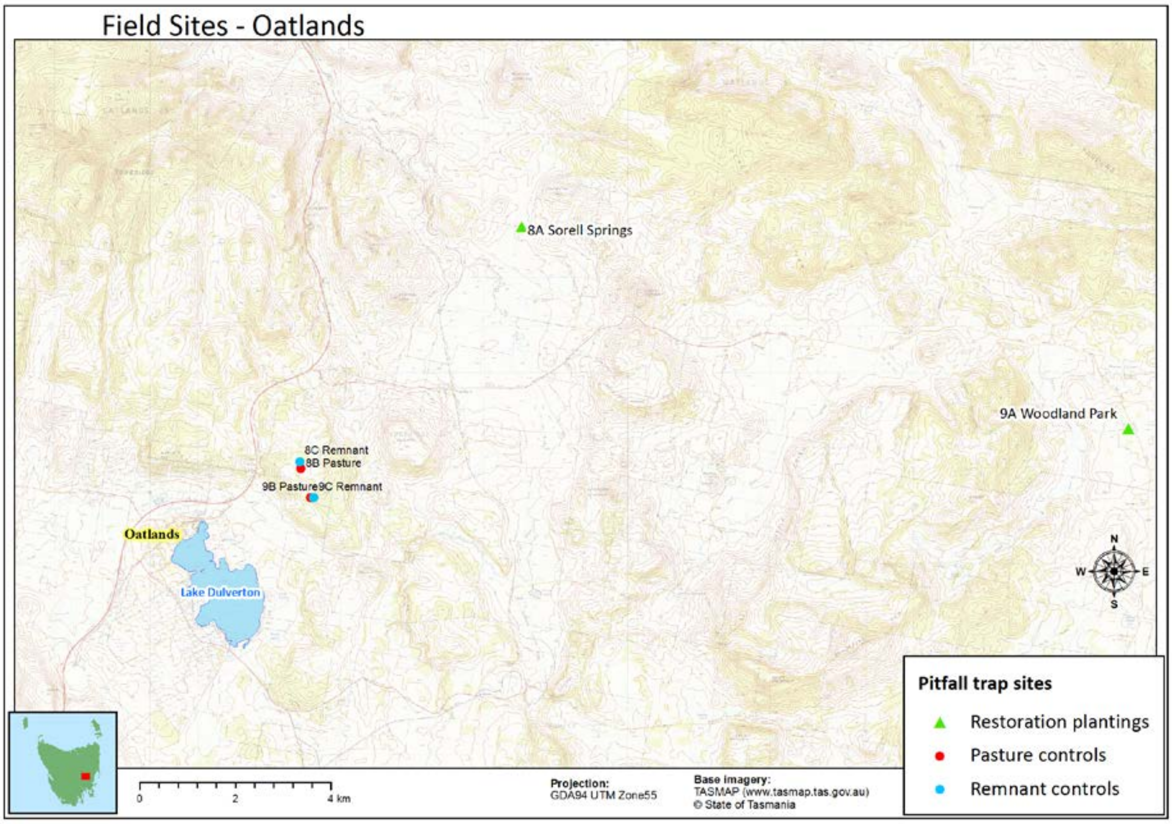

Figure 2 continued.

3.3 Restoration plantings

3.3.1 Site 1A: "Meadowbank" (-42.6389°S, 146.8214°E).

The site at Meadowbank is a 17.8 ha planting on an eastern facing slope at an elevation of 260m to 300m above sea level. It is bordered by remnant woodland patches on 2 sides, and slightly wooded pastures on the uphill and downhill slopes. The planting was established in 2011 and consists of even rows of *Eucalyptus pauciflora* and *Eucalyptus tenuiramis* in a Triassic sandstone soil. An understory of *Acacia dealbata* was present, with a sparse ground cover of approximately 15% bracken fern *Pteridium esculentum* and 5% scotch thistle. The ground was estimated to be 80% bare. The traps in this planting were set approximately 200m from a nearby remnant woodland patch.

Whilst the Meadowbank planting was fully fenced during our first sampling, a severe bushfire swept through the property shortly afterwards, bringing down trees and destroying fences. Whilst the planting itself was untouched by the fire, sheep were then able to access the planting via either fence damage or gates left open by emergency services personnel. By the time we returned in autumn it was clear that sheep had been able to browse extensively throughout the planting. As a result, the undergrowth was largely trampled and the eucalypt saplings were mostly decimated.

3.3.2 Site 2A: "Olympic Landcare" (-42.5406°S, 146.7723°E).

This site is situated at Jericho and managed by the Olympic Landcare group. It is a 0.6 ha planting measuring approximately 40m wide by 140m long. The southern edge is bordered by the Lyell Highway with pastures bordering the other three sides. There is a small dam to the north. The planting is positioned on flat land at 113m above sea level. The planting was established in 2000 and consists primarily of *E. pauciflora* and *Acacia verticillata*, with a varied shrubby understory of *Bursaria spinosa*, *Banksia marginata*, *A. dealbata* and *Acacia melanoxylon*. The soil is silty loam on sandstone bedrock. The ground cover consists of an estimated 50% mixed grass species which are mostly escaped pasture grasses, as well as a 30% covering of leaf litter, and approximately 20% bare earth with a scattering of lichen, moss, and rocks. The nearest remnant patch is on the other side of Lake Meadowbank, a large impoundment measuring approximately 130m from one side to the other at the closest point of crossing. The traps were set approximately 3100m from this remnant woodland patch.

3.3.3 Site 3A: "Uralla" (-42.5462°S, 146.8493°E).

The site is a 2.7 ha planting measuring approximately 220m by 100m, on a western facing slope at an elevation of 115m to 130m above sea level. It is bordered by pastures on two sides, with a remnant woodland patch bordering the southern and eastern edges. Established in 1998, the planting is a direct seeded regeneration with *Eucalyptus globulus*, *E. pauciflora* and *E. delegatensis* planted in rows across the slope. An understory of *A. dealbata* and *Dodonaea viscosa* was present. The ground cover consists of mostly mixed grasses species both native and exotics, with 20% leaf litter, 10% moss and stones, and 20% bare earth. The soil is sandy loam on dolerite bedrock. The traps were set 80m from a remnant patch.

3.3.4 Site 4A: "Dungrove" (-42.2688°S, 146.8872°E).

This site is a 31.1 ha planting on a very gentle western facing slope, at an elevation ranging from 550m to 570m above sea level. It is bordered by a pasture on the downhill / western edge, with remnant woodland on all other sides. Established in 2011, it consists exclusively of *E. pauciflora* and does not exhibit an understory. The ground cover consists of about 30% mixed pasture grasses and the graminoid *Lomandra longiflora*, with 10% broadleaf weeds, 10% rocks and moss and lichen, and 50% bare. The soil type is sandy loam.

Figure 3. Damage to the trunks of young saplings in Planting 4A caused by wild fallow deer (*Dama dama*).

This site is regularly intruded by wild fallow deer (*Dama dama*) and so the eucalypt planting exhibits signs of grazing including damage to bark on their trunks (see Figure 3). The traps were set 300m from the nearest remnant patch.

3.3.5 Site 5A: "School Block" (-42.3869°S, 147.0170°E).

This is the only peri-urban planting within the study and exists on a 100m x 165m flat block (1.6 ha) within Bothwell township, bordered by roads on two sides and by fencing and pasture on the other two sides. The elevation is 367m above sea level. Established in 1994, it consists of *Eucalyptus amygdalina*, *E. viminalis*, and *E. pauciflora* planted in straight rows in silty loam on sandstone bedrock. There is no understory, and there is a thin ground cover of approximately 75% leaf litter, 20% native grasses, and 5% bare soil. There are few weeds. The nearest remnant patch is 950m away.

3.3.6 Site 6A: "North Stockman" (-42.4965°S, 147.1973°E).

This planting, established in 1988, is the oldest in the study. Originally a 5ha T-shaped planting measuring 600m at its greatest width, the centre of the planting was recently destroyed to make room for cropping, forming two separate plantings. The portion which was chosen for trapping is approximately 4.5 ha and is bordered by pasture on three sides and a poppy crop on the downhill side. The site is on a western facing slope with elevation ranging from 230m to 260m above sea level. The planting is a mixture of *E. pauciflora, E. tenuiramis* and *A. dealbata* with an understory of *Bursaria spinosa* and *Banksia marginata*. The ground cover is estimated as 75% pasture grasses, 20% leaf litter and approximately 5% bare earth. The soil type is silty loam. The nearest remnant patch is 1750m away.

3.3.7 Site 7A: "Grassy Hut" (-42.3985°S, 147.0803°E).

Established in 2011, this planting is approximately 330m by 460m at its widest point, with an area of 17.0 ha. The site is a gently sloping north face at an elevation ranging from 440m to 460m above sea level. The planting consists of *E. pauciflora* and *E. tenuiramis* with an understory of *A. dealbata*, in silty loam soil on mudstone bedrock. The ground cover is approximately 70% pasture grasses, with a high weedy presence and about 20% bare earth.

The site is bordered by pasture on three sides, with a sparse remnant patch to the south. The traps were set 660m from the remnant patch.

3.3.8 Site 8A: "Sorell Springs" (-42.2422°S, 147.4595°E).

This T-shaped planting has a distance of 450m across its top, and 650m from top to bottom. The width of the planting is 30m throughout, giving it an area of 2.8 ha. It is bordered by pasture on all sides. It is a near flat patch of land at an elevation of 380m. Established in 2003, the planting consists of exclusively *E. pauciflora*, with an understory of *A. dealbata*. The ground cover is a mix of native and exotic grasses covering approximately 30% of the ground, a 20% coverage of leaf litter, with the remaining 50% being bare earth. The soil type is silty loam. The nearest remnant patch is 1000m away.

3.3.9 Site 9A: "Woodland Park" (-42.2795°S, 147.6139°E).

The planting site has a flat aspect at an elevation of 396m. This experimental planting has a total area of 2.7 ha but within this, a number of different species groups have been planted in their own blocks. The total planting is a narrow L shape measuring 330m at its north-south length, and 560m along the east-west line, with a width of approximately 20m along its entire length. Rows of mature *Pinus radiata* trees divide the planting from neighbouring paddocks, which are used for cattle grazing. The traps were set within an *E. pauciflora* block measuring 20m across and 140m long, ie a planting block of 0.28ha within a total planting area of 2.7 ha. The planting has a mixed understory of *Dodonaea viscosa* and *Bursaria spinosa*, with the ground cover made up of an estimated 40% exotic grasses, 50% leaf litter, and 10% bare earth. The soil type is silty loam. The nearest remnant patch is 1960m away.

3.3.10 Site 10A: "Curringa" (-42.5638°S, 146.7780°E).

This planting measures 220m x 20m (0.5ha) on a flat parcel of land at an elevation of 119m above sea level. It is bordered by pastures on all sides. Established in 1993, it is the only non-eucalypt-dominant planting in the study, and consists of *Dodonea viscosa, Acacia melanoxylon,* and *Eucalyptus leucoxylon*. The understory is *A. dealbata*. This planting has a shrubby ground cover, with *Epacris* spp quite prominent. The remaining ground cover is an estimated 50% leaf litter, 20% pasture grasses, 30% bare earth, and a small amount of stones, lichen and moss. The soil type is silty loam on dolerite bedrock. The nearest remnant patch is 1000m from the site of the traps.


1A Meadowbank


2A Olympic Landcare


3A Uralla


4A Dungrove


5A School Block


6A North Stockman

Figure 4. Photos showing environmental condition of each restoration planting site from 1A to 6A.

7A Grassy Hut

8A Sorell Springs

9A Woodland Park

10A Curringa

Figure 4 cont. Photos showing environmental condition of each restoration planting site from 7A to 10A.

3.4 Control sites

Nine pasture control sites were selected, each in separate pastures adjacent or near to a restoration planting. All are long-established working sheep paddocks which have been cleared and sown with exotic pasture grasses. All paddocks displayed a variety of herbaceous weeds including *Medicago* spp, *Rumex* spp, dock, and a variety of other broadleaf weeds. Scotch thistle *Onopordum acanthium* was observed in all but one of the pasture control sites. The pastures chosen are typical of sheep paddocks in the area and displayed similar environmental characteristics such as altitude, climate, rainfall, as the restoration planting to which they were being matched.

Nine woodland control sites were established in nearby native remnant patches. These were selected to show typical environmental characteristics to the planting sites to which they were matched, including altitude, climate, and rainfall. They displayed a variety of vegetation types, ground cover, and fire histories and varied greatly in their distance to a planting.

This research design resulted in ten restoration planting sites, nine pasture control sites, and nine remnant woodland patch control sites for a total of 28 sites.

3.5 Pitfall traps

Pitfall traps were the sampling tool chosen as they effectively target ground-dwelling invertebrates. They are an appropriate method for medium term studies with the goal of monitoring land use decisions such as logging, grazing, prescribed burns and restoration (Underwood & Fischer 2006).

Plastic cups of 9cm diameter and 15cm depth and with a volume of 200mL, were set into the ground as pitfall traps, using a soil corer and a small hand-held trowel to dig the holes, taking care to avoid excess disturbance to surrounding soil. Each pitfall trap consisted of two cups, one set inside the other so the inner trap could be lifted and emptied with minimal disturbance to the surrounding soil. Each inner cup was filled with approximately 2cm of propylene glycol as a preservative. Different preservatives can be used in pitfall traps including ethanol, glycerol, ethylene glycol, saline solutions, detergent, propanol, and water, and various combinations of the above. In this study, propylene glycol was chosen due to its odourless properties, making it neither attractive nor repellent to insects, and because of its non-toxicity to mammals (Underwood & Fischer 2006).

At each sampling site, three pitfall traps were set in a 15m transect with 5m spacing between each. “Hard” or “sharp” edges, such as those created by a fenced plantation bordering a pasture, can markedly impact the distribution of fauna in a landscape (Dauber & Wolters 2004) so to reduce the impact of edge effects, each transect was run parallel to the edge of the vegetation type and at a minimum distance of 10m from the edge. In the summer sampling cycle, all traps set in pastures were covered with a 12.5cm square cover made of 12mm steel mesh. These covers were designed to protect livestock from injury while protecting the trap from potential damage caused by animal disturbance (see Figure 5). It was found that these covers reduced the amount of by-catch such as lizards and frogs falling into traps, and also greatly reduced the amount of dirt and leaf litter entering the traps which could potentially aid specimens to escape the trap. A reduced amount of dirt and detritus in the traps also reduces processing time. As such, in the autumn sampling cycle it was decided to put covers over all pitfall traps in all vegetation types.

Figure 5. Uncovered pitfall trap in site 5C (L) and pitfall trap in site 1A with 12mm steel mesh cover (R)

The total number of traps laid was 84 in summer (30 in restoration plantings, 27 in pastures, and 27 in remnant woodland), and 81 in autumn (27 in each vegetation type) for a total of 165 traps.

During the January sampling cycle, 36 of the traps were left in situ for 7 days, and 48 traps were left for 6 days, before all being collected on the same day. Upon collection, four traps were found to be disturbed. Contents of the traps were immediately transferred into clean specimen jars, rinsed with ethanol, and sealed for transportation and storage.

This process was repeated in the autumn sampling cycle, however the traps were left in place longer due to the expected lower level of invertebrate activity during shorter day length and autumn weather, despite relatively warm weather during the time period including one day where the temperature reached 24 degrees. Of the 81 traps set in the autumn cycle, 60 were collected nine days after setting the traps, with the remaining 21 traps collected after 11 days.

Two traps were disturbed – one in a restoration planting and one in a remnant patch – leaving 26 good samples from restoration plantings, 26 from remnant patches, and 27 from pastures.

The contents of each trap were transferred in the field into sealed sample jars with 75% ethanol solution and transported to the laboratory. They were then sorted into taxonomic groupings of ants, beetles, spiders and "other," ready for further identification. Samples were then identified to species level where possible or assigned to morphospecies. All data was adjusted to a 10 day standardisation.

Voucher specimens of the material were archived in glass vials with 75% ethanol, and deposited at the UTAS School of Geography & Environmental Studies collection for future reference. Each vial bears a label on acid-free paper providing full details of locality, date, and treatment, as well as catalogue number. A full catalogue listing is given in Appendix 12.

3.6 Seasonality

Two sampling cycles were undertaken, a summer cycle in January 2013 and an autumn cycle in April 2013. The aims of conducting a second cycle were:

a) To account for seasonality in invertebrate activity
b) To include a greater range of species in the analysis
c) To expand the dataset and improve resolution in the results

The autumn trapping cycle was conducted at the same sites as the summer cycle, and used the same number of traps at each site. In most cases the original trap casing set into the ground in summer was relocated in autumn and reused. One planting site (6A North Stockman) was not included in the autumn cycle after access was cut off due to a new poppy plantation surrounded by an electric fence. The total number of traps laid was 84 in summer (30 in restoration plantings, 27 in pastures, and 27 in remnant woodland), and 81 in autumn (27 in each vegetation type) for a total of 165 traps.

The was one minor difference in the methodology when repeated in autumn, in that the summer cycle steel mesh covers were placed on all the traps in pasture sites only, whereas in the autumn cycle these covers were placed on all traps in all treatments.

Habitat changes due to seasonal variation, or from one year to the next due to large-scale climate patterns, may influence invertebrate activity and therefore affect the results. The effect that weather has on vegetation and water quality is clearly visible in Figure 6, which shows how the condition of a habitat can change over the course of two years.

(a)

(b)

(c)

Figure 6. Google Earth imagery of site 2A Olympic Landcare, showing seasonal differences and annual variations. Photos are dated: (a) 15 January 2008 (b) 18 November 2009 (c) 8 February 2010

3.7 Environmental variables

The five environmental factors which were introduced in the methods – soil clay content, planting area, elevation, proximity to remnant patch, and ground cover – were analysed against the abundance and species richness data to determine whether any relationships exist. The data analyses for soil clay, elevation and ground cover were conducted on data taken from all sites in all treatments. The analysis of proximity to remnant patch was undertaken with data from the plantings and pastures only. The analysis of the planting area was undertaken with data only from the plantings. In addition, abundance and species richness data was analysed against the age of the planting, using data only from the sites in the plantings, not the controls.

Percentage cover was estimated for each of the ground cover variables grasses, leaf litter, bracken and shrubs, rocks / moss / lichen, bare ground, and weeds. Any of these variables could be correlated with another, lessening the legitimacy of using all these considerations together. A multivariate analysis of these components showed none were significantly correlated to any other, allowing the freedom to use all variables in the tests.

A full list of the environmental variables recorded for this study is shown in Table 2, and a summary of records for all planting sites can be found in Appendix 2.

3.7.1 Soil clay content

Data on soil type and % clay at each planting was sourced using the Australian Soil Resource Information System (www.asris.csiro.au/mapping, accessed 9 Sept 2013).

Soil chemistry and texture has a strong association with resident ant communities (Boulton *et al* 2005). Many studies have found relationships between soil clay levels and the relative abundance of a particular ant species. For example, the Argentine ant *Linepithema humile* prefers clay loam over sandy soils (Way *et al* 1997), and there is a correlation between clay content and the relative abundance of the desert seed-harvester ants *Pogonomyrmex rugosus* and *Messor pergandei* (Johnson 1992). The presence of clay in soil can influence other causative factors that affect relative abundance of a particular invertebrate species. Soils with heavy clay components demonstrate lower water absorption rates and may also react with minerals in the soil such as sodium and potassium to form carbonates (Emerson 1966).

3.7.2 Planting area

The area of each planting was measured using Google Earth imagery.

In a fragmented habitat such as the ones in which this study was undertaken, patch area and in turn the amount of patch edge can have an important effect on the distribution of ground-dwelling invertebrates (Golden & Crist 2003). A hard edge between habitats can affect species' movements across the habitat boundary. Larger sized patches and plantings will generally have a lower ratio of boundary to area, depending on shape. The exceptions are two narrow shelter-belt Plantings 8A and 9A, which are only 30m and 20m wide respectively, giving both a very high boundary-area ratio. Revegetation projects should ideally be conducted in patches that are large, wide and structurally complex to maximize the benefits to fauna (Munro *et al* 2007).

3.7.3 Elevation

Elevation was measured using a handheld Garmin GPS device in the field and checked against Google Earth imagery.

Species richness for some invertebrate taxa increases at higher elevations, whereas for other taxa diversity is greater at lower elevations where there is usually a higher level of primary production. Ant species richness is known to vary with altitude, with a common pattern being a peak in species richness mid-gradient, followed by an exponential decline as elevation rises (Brühl *et al* 1999; Sanders *et al* 2003; Botes *et al* 2005). The important factor is that an altitudinal gradient usually coincides with changes in mean temperatures, annual rainfall, and changes in the dominant vegetation. These variables can explain up to 80% of the variation seen in ant species richness across an altitudinal gradient (Sanders *et al* 2003).

3.7.4 Proximity to remnant patch

The Euclidean distance from the location of the traps in each planting to the nearest remnant patch was measured using Google Earth imagery.

The degree of isolation of a planting may have an impact on recolonisation in terms of species richness, but also community structure. Andersen (1993) found that rehabilitated minesites had considerably higher ant species richness when the sites were located close to relatively undisturbed vegetation.

3.7.5 Ground cover and weediness

Percentage ground cover was estimated using a standard 1m x 1m quadrat placed at each site at one randomly selected location, and through visual estimates in the field and cross checked against photographic evidence. Ground cover was estimated for the six categories Grasses, Leaf Litter, Bracken or Shrubs, Rocks/Moss/Lichen, Bare, and Weeds and assigned a percentage value for each. The presence or absence of scotch thistle was also recorded as a particular case of weediness.

Whilst plants generally impact ant species richness and abundant to a lesser degree than soil chemistry, the abundance of certain invertebrate taxa can be readily correlated with plant biomass or plant species richness (Boulton *et al* 2005). Variations in ant species richness can be correlated with a range of factors such as ground herb cover, soil moisture, plant species richness, foliage cover, presence or absence of bare ground, and the amount of leaf litter (Majer 1985; Lassau & Hochuli 2004). Soil moisture and litter biomass is a major contributor to not just abundance and species richness, but also to the composition of an ant community in terms of functional groups (York 2000). Specific to restoration plantings, litter cover and the disappearance of bare ground over time strongly influence changes in ant species richness (Majer 1985; Andersen 1993).

Table 2. Identity, type and range of variables used in this study.

Variable	Type	Range
Treatment	Categorical	Pasture, Planting, Remnant
Latitude (°S)	Numerical	42.2422 - 42.6395
Longitude (°E)	Numerical	146.7723 - 147.6139
Plantation established (Year)	Numerical	1988 – 2011
Plantation age (years)	Numerical	2 – 25
Plantation area (ha)	Numerical	0.5 - 31.1
Elevation (m)	Numerical	102 – 580
Monocalyptus (incidence)	Categorical	0, 1
Symphyomyrtus (incidence)	Categorical	0, 1
Allocasuarina (incidence)	Categorical	0, 1
Acacia (incidence)	Categorical	0, 1
Grasses (% cover)	Percentage	0 – 95
Leaf litter (% cover)	Percentage	0 – 90
Bracken or shrubs (% cover)	Percentage	0 – 80
Rocks / moss / lichen (% cover)	Percentage	0 – 50
Bare (% cover)	Percentage	0 – 80
Weeds (% cover)	Percentage	0 – 25
Scotch thistle (incidence)	Categorical	0, 1
Parent geology	Categorical	sandstone, basalt, dolerite, mudstone
Fertility of bedrock (rank)	Ordered	1, 2, 3
Soil clay (%)	Percentage	15 – 30
Proximity to remnant patch (m)	Numerical	20 – 3100

3.8 Data Analysis

Rank abundance and species richness for each taxonomic group (ants, beetles and spiders) were calculated, and summarised for the summer and autumn cycles separately, and collectively. The dataset for the spider group was found to be inadequate, and was not used in this study. However, a full list of the spider species/morphospecies collected in this study, with abundance counts, has been included in Appendix 5, for completeness.

The dataset of ants and beetles was used to compare total species richness and abundance between the three habitat types and for the analysis of assemblage composition and indicator taxa for both habitats. The species composition over two seasons was analysed in order to note any significant differences.

For the rank abundance, the data for the planting sites was standardised to account for the uneven numbers of treatment sites. All summer data was then standardised to a 10-day figure to match up with the autumn data. Relative abundance for all species was arranged in order of rank, and plotted graphically for demonstration.

3.8.1 Abundance

The counts of ant taxa were analysed using JMP (SAS Institute Inc. 2012) to determine whether they followed a normal distribution. A standard one way analysis of variance was used to test for significant differences in abundance between the three treatments.

The total abundance of each taxon was tabulated from the data for each season. Species were sorted in descending order according to total abundance and then summarised in rank-abundance bar graphs. An expected result in biologically diverse communities is that some taxa are present at very low abundance and can be indicative of a variety of ongoing processes (i.e. indicators of truly rare species in the sampled habitat or accidently occur as migrating or vagrant species). Rare species (individual species abundance <0.5% of the total) were removed from the dataset for some comparative analyses in order to facilitate extraction of the main trends and community patterns. The species richness data was analysed using JMP to confirm a parametric distribution (see Appendix 6). A Tukey-Kramer HSD test was then used to determine whether there were significant differences in the mean values for the three treatments.

3.8.2 Community data

The relationships between invertebrate assemblages at the sample sites were explored and displayed in ordinations generated by non-metric multidimensional scaling (NMDS). Abundance data was log (x+1) transformed before analysis to down weight the influence of very common species which could skew the results. The Sorenson Index was used as the measure of dissimilarity for pairwise comparisons and the default options employed within PC-ORD (McCune & Mefford 1999). Suitably low stress values were usually achieved in 2 or 3 dimensions. Environmental variables which significantly correlated with the ordination were plotted as vectors in the ordination space. A cut-off value of $r^2 = 0.2$ was used.

A Multi-Response Permutation Procedures (MRPP) test which is a non-parametric procedure for testing the hypothesis of no difference between two or more groups of entities, was used to test the hypothesis of no difference between the sites during different seasons (McCune & Mefford 1999). This has the advantage of not requiring assumptions (such as multivariate normality and homogeneity of variances) that are seldom met in ecological data of this sort. The MRPP statistic delta is simply the overall weighted mean of within-group means of the pairwise dissimilarities among sampling units.

Further multivariate of analysis was used to extract meaning from the data. Sample sites were ordinated using PC-ORD software (McCune & Mefford 1999) on the basis of their invertebrate fauna. Ordination is a multivariate analytical method that arranges sampling units along axes such that similar sites are plotted close together and dissimilar sites are further apart. The result is an objective summary of the relationship between sampling units in a low dimensional species space. The goal is to reveal underlying structure in the data that represent patterns of species occurrence as determined by environmental variables. The Non-metric Multidimensional Scaling (NMDS) used in this study is an ordination method that is well suited to data that are non normal or are on arbitrary, discontinuous, or otherwise questionable scales. NMDS is generally regarded as the best ordination method for community data (Faith *et al* 1987). A Monte Carlo test of significance was included.

3.8.3 Spatial issues

Due to the large distance (kilometres to tens of kilometres) separating some of the sites, it is possible that invertebrate communities may be affected by spatial relationships, i.e. biotic communities close together in space may be more similar than those located further away independent of environmental factors.

The Mantel test in the ecodist package in R 2.2.1 (Goslee & Urban 2007), was used to test whether the distance matrix based on invertebrate communities was correlated with the Euclidean matrix based on geographical distance (derived from Eastings and Northings). P-values were for the two-tailed test and were based on 10,000 permutations.

3.8.4 Correlation among environmental variables

The correlation between habitat variables was tested using Spearman rank correlation. Where pairs of variables have a correlation coefficient that is high (eg > 0.8), one of the pair is randomly excluded from further analysis (Milne *et al* 2006) to seek the most parsimonious combination of descriptors for each species. A generalised linear model (GLM) was used to seek a predictive distribution model based on species-habitat relationships (Milne *et al* 2006).

3.8.5 Indicator values

The practice of using "indicator species" in environmental monitoring is a somewhat contentious issue which some ecologists believe is misleading or inappropriate (Landres *et al* 2005) and one that should be applied with caution (Carnigan & Villard 2001). McGeogh (1998) defines biological indicators as "*a species or group of species that readily reflects: the abiotic or biotic state of an environment; represents the impact of environmental change on a habitat, community or ecosystem; or is indicative of the diversity of a subset of taxa, or of wholesale diversity, within an area*." Their application in this study therefore seems wholly appropriate.

"Indicator values" are used to demonstrate which species assemblages are typical for a particular habitat, and add ecological meaning to groups of sites. Identifying which species, if any, are indicative of a particular environment can be a valuable tool for conservation and for understanding the ecological values of a habitat. Simply recording a high species diversity at a site isn't necessarily indicative of high ecological value; likewise, an environment with low species diversity may in fact support a high proportion of rare or threatened species.

An Indicator Species Analysis (Dufrêne & Legendre 1997) provides a simple, intuitive solution to the problem of evaluating species associated with groups of sample units. It combines information on the concentration of species abundance in a particular group and the faithfulness of occurrence of a species in a particular group. It produces indicator values for each species in each group which are tested for statistical significance using a Monte Carlo technique.

3.9 Taxonomic sufficiency

Specimens collected in this study were identified using taxonomic literature (Andersen 1991) and the CSIRO website Atlas of Australian Ants http://anic.ento.csiro.au/ants, as well as the assistance of taxonomic specialists to assist identification where appropriate. A high level of entomological expertise is generally required to accurately identify the hundreds of insect species typical of this type of research. This can potentially cost a large amount in consulting fees, if external expertise is called upon. Alternatively, researchers may develop enough knowledge to identify a species to a genus or "morphospecies" level, which is a level of taxonomic resolution sufficient to distinguish one species from another in an analysis. For this study, where the aims are to compare the abundance and species richness of invertebrates in different treatments, morphospecies identification has been shown to be taxonomically sufficient (Schnell *et al* 2003, Brennan *et al* 2006). Specific to ant research, another taxonomically sufficient approach is to employ a functional group classification, which considers competitive interactions and habitat requirements predominantly at the genus level (Andersen 1995).

Chapter 4 Results

4.1 Ants

This study collected a total of 2589 ants, comprised of 32 morphospecies representing 19 genera. A summary of abundance per treatment is given in Table 3, with complete species list available in Appendix 3.

4.1.1 Ant abundance

Ants were clearly much more active during the summer, with twice as many ants being collected in January compared to later in April. The summer trapping cycle collected 1770 ants from 28 species, with the autumn trapping cycle collecting 819 ants from 26 species. Ant abundance was highest overall in the plantings, with 1110 ants or 42.9% collected from the 10 plantation sites, 879 or 39.9% collected from the 9 remnant patches, and 600 ants or 23.2% collected from the 9 pasture sites.

Table 3. Ant abundance per treatment in each cycle.

Treatment	**Summer**	**Autumn**	**Overall**
Pasture	405	195	600
Plantings	866	244	1110
Remnant	499	380	879
TOTAL	**1770**	**819**	**2589**

The ten most abundant species made up 87.0% of the total population collected. Another ten species were present in counts of 5 individuals or less after both trapping cycles, with 4 of these species collected as singletons.

The most abundant species overall was the seed harvesting ant *Pheidole tasmaniensis* with a count of 467, which was 18.1% of all ants collected. The largest number of this species found in any one trap was 65 individuals (trap 3Bi from the summer cycle), and counts greater than 30 were collected in five different traps. Whilst this suggests that the data may have been affected by nearby nesting sites, it also suggests that *P.tasmaniensis* nests are relatively common. In fact, this species was found to be abundant across all treatments and at most sites, being recorded in 20 of the 28 trapping sites in either the summer or autumn cycles.

Four species were found in all three treatments at both trapping cycles – *Pheidole tasmaniensis*, *Myrmecia pilosula*, *Rhytidoponera tasmaniensis* and *Rhytidoponera victoriae*. These four species were amongst the 5 most common by rank abundance, with *Iridomyrmex bicknelli* making up the fifth. This latter species was found in all treatments in summer and in two treatments in autumn.

Five uncommon species occurred only in January (*Ochetellus* sp., *Epobostruma* sp., *Prolasius nitidissimus*, *Melophorus* sp. and *Stigmacros* brown) and another five taxa were recorded only in April (*Cerapachys* sp., *Amblyopone australis*, *Myrmecia esuriens*, *Monomorium* black and *Monomorium* large).

The rank abundance curve for ants (Figure 7) shows a profile typical of invertebrate communities. Typically, some species are common, and most species are uncommon, resulting in a long tail of species recorded in very small numbers.

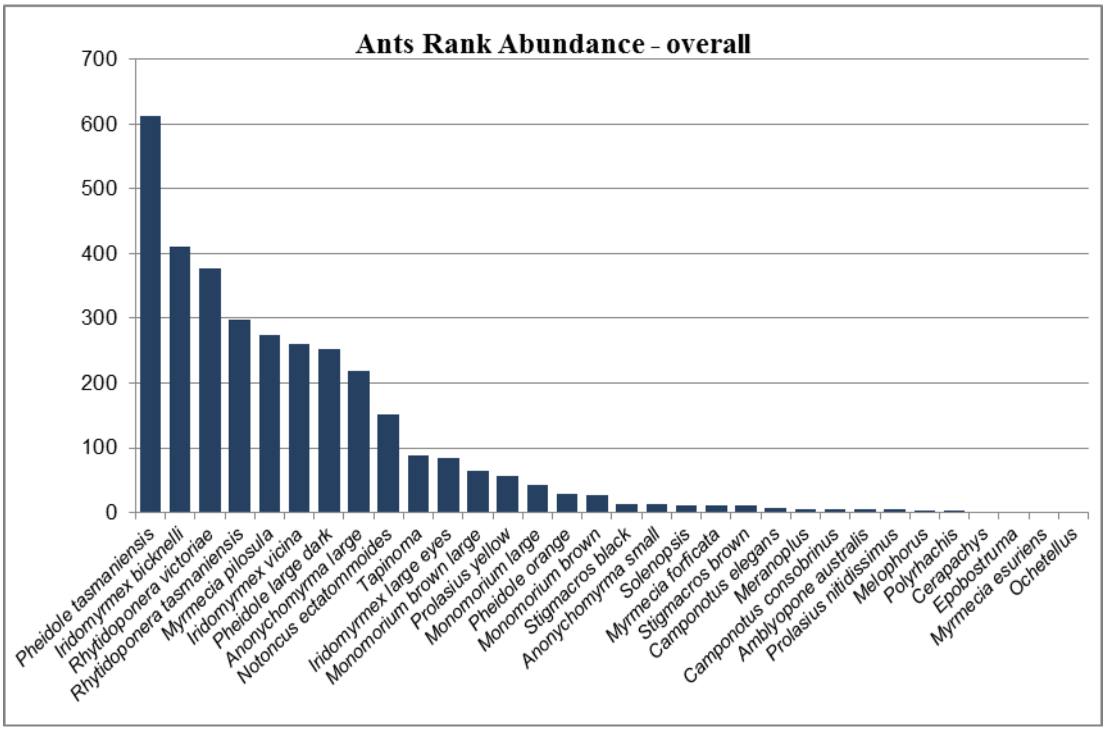

Figure 7. Ants Rank Abundance across all treatments for both seasons combined. It was assumed that all traps were in situ for 7 days during the summer cycle and figures for both seasons were then standardized to a 10 day average.

The frequency distribution for ant abundance had a non-parametric distribution (see Appendix 6), and mean counts between treatments were found to be not significantly different ($F_{2,53}$=0.9157, p=0.4065). There is however a clear progression with numbers in the plantings 85% higher than that in the pasture on average (see Figure 8).

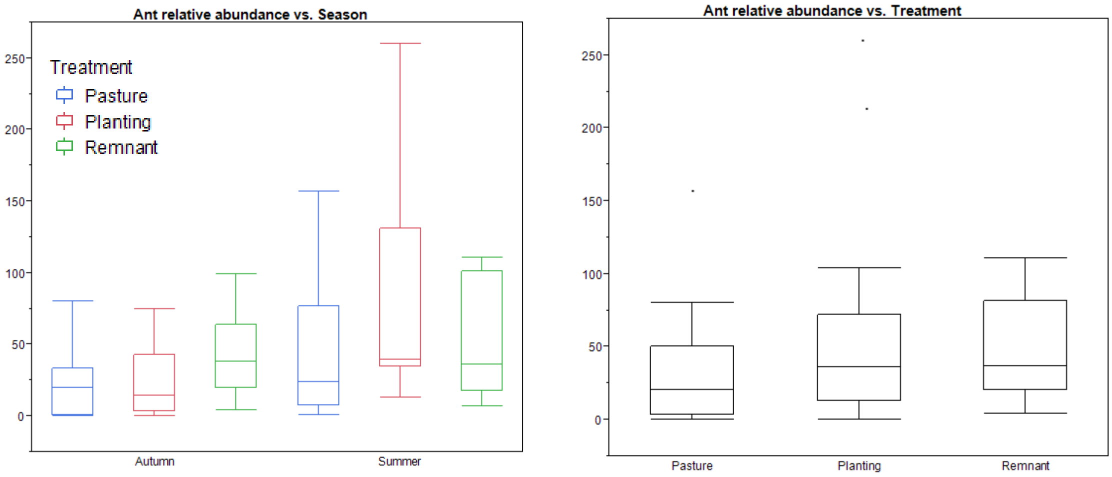

Figure 8. Ant abundance per treatment in the summer and autumn cycles, and overall abundance per treatment. Differences between treatments are not significant ($F_{2,53}$=0.9157, p=0.4065)

4.1.2 Ant species richness

Ant species richness was highest in the remnant patches with 28 species recorded in remnant woodlands, 9 of which were unique to this treatment. The restoration planting treatments supported 19 species, of which 3 were unique and not found in other treatments. The pastures yielded 14 species overall, all of which were found also in either the remnant woodlands, or the plantings, or both. Nine species were common to all three treatments.

Whilst the mean species richness does increase from the pasture sites to the planting sites in both seasons, the difference in species richness between the treatments was not statistically significant ($F_{2,53}$=2.67, p=0.079) (Figure 9).

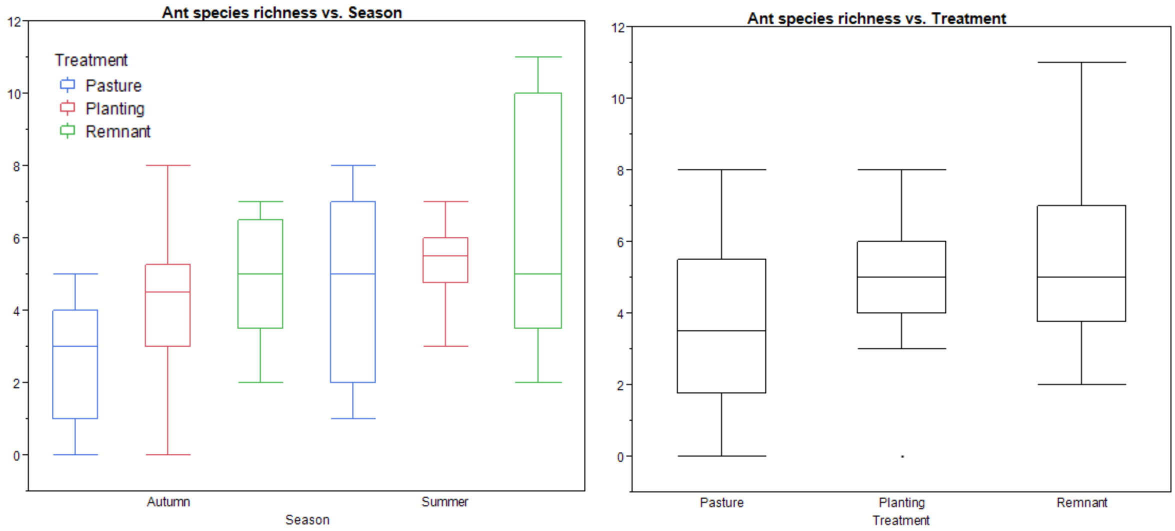

Figure 9. Mean of ant species richness per site for each treatment in both seasons, and for both seasons combined, with a standard error of one. There is no significant difference in the overall mean species richness between the pasture and planting sites nor between the planting and the remnant ($F_{2,53}$=2.67, p=0.079).

4.1.3 Ant communities

In both seasons, non-metric multidimensional scaling (NMDS) ordination of the sites (Figures 10 and 11) shows that the remnant vegetation sites possess the greatest variation in ant communities. The polygons represent community groupings for each treatment in ordination space and the polygon for the remnant sites is clearly larger than the other two treatments. Additionally, the communities in the remnant vegetation samples largely occupy ordination space separate from the other habitats, not overlapping at all with the other two treatments in summer, and only moderately overlapping with the plantings in the autumn ordination. The ant communities of the pasture and plantation sites are somewhat intermingled, as indicated by the moderate degree of overlap between their polygons for both seasons.

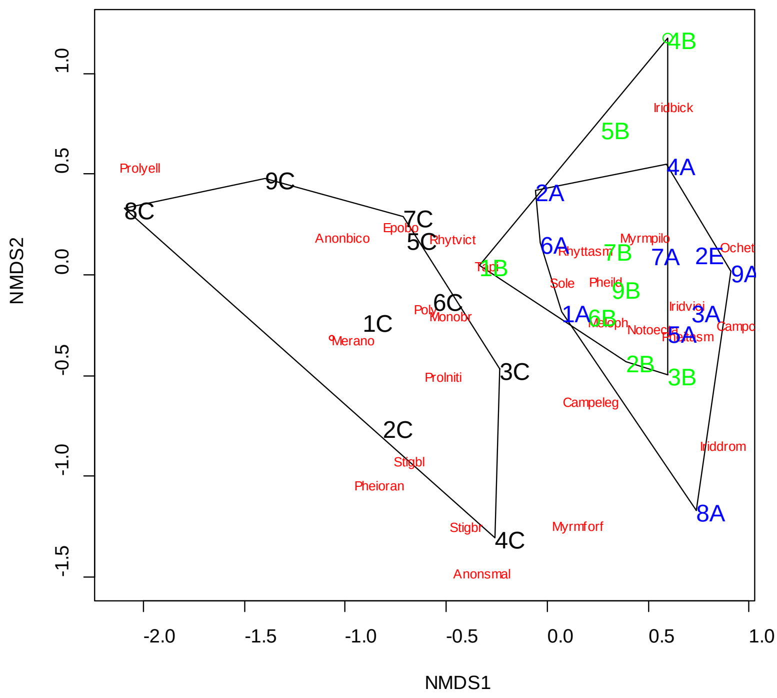

Figure 10. Ordination (NMDS) of the sites on the basis of their ant fauna sampled in January 2013 (summer). Ant counts were square root transformed before analysis. Dissimilarity measure was Bray-Curtis. Stress in 2D = 0.1497. A=planting, B=pasture, C=remnant vegetation. Polygons connect the treatments.

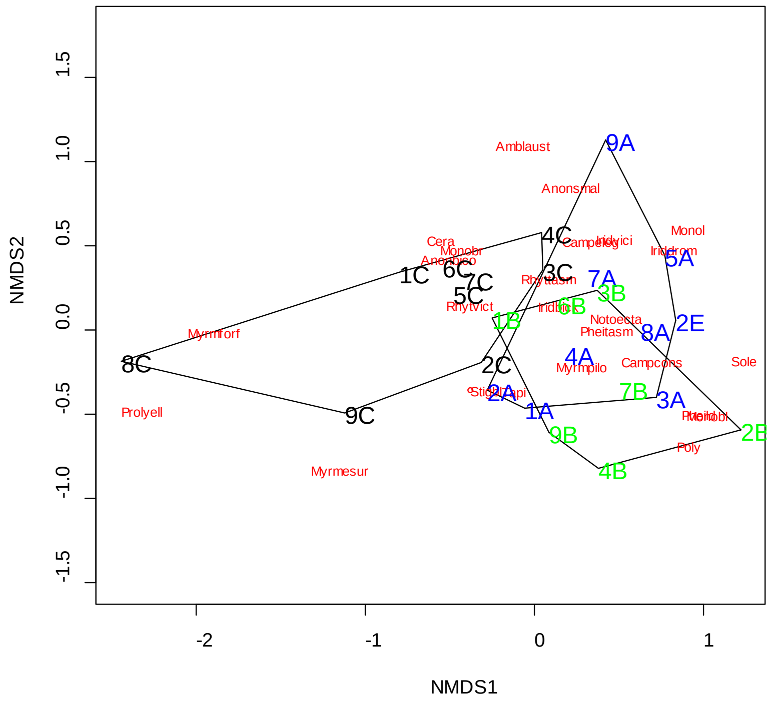

Figure 11. Ordination (NMDS) of the sites on the basis of their ant fauna sampled in April 2013 (autumn). Ant counts were square root transformed before analysis. Dissimilarity measure was Bray-Curtis. Stress in 2D = 0.1610. A=planting, B=pasture, C=remnant vegetation. Polygons connect the treatments.

4.1.4 Indicator values

Only a small number of ants qualified as indicator species and there were differences in indicators between seasons (Tables 4 and 5). Two species were found to be indicative of the remnant woodland patches in both seasons. These were *Anonychomyrma biconvexa* which was a consistent indicator, and *Rhytidoponera victoriae* which was a weaker indicator of remnants in the summer. For the pastures, *Iridomyrmex vicina* remains active in this treatment until April, when it becomes indicative of this land use type. Remnant vegetation supported the greatest number of indicators: *Meranoplus* in summer, *Rhytidoponera victoriae* and *Monomorium* brown in autumn and the largely tree-dependent *Anonychomyrma biconvexa* in both seasons. The thermophilic *Iridomyrmex vicina* was indicative of the pasture habitat in autumn. No species were found to be indicative of the plantings.

Table 4. Indicator values for ant taxa collected in the summer sampling cycle. Ants counts were square root transformed. Freq is the number of times a species was present among samples (i.e. not its abundance). Significant indicator taxa are highlighted. ns = not significant; * = p<0.05 ; ** p<0.001 .

Ant code	Ant species	Treatment	Indicator Value	Freq	p value	Sig.
Rhyttasm	*Rhytidoponera tasmaniensis*	Planting	0.7037	19	0.452	ns
Pheitasm	*Pheidole tasmaniensis*	Planting	0.5185	14	0.818	ns
Iridvici	*Iridomyrmex vicina*	Planting	0.4074	11	0.962	ns
Myrmpilo	*Myrmecia pilosula*	Planting	0.3704	10	1	ns
Iriddrom	*Iridomyrmex dromus*	Planting	0.1852	5	1	ns
Monobr	*Monomorium* brown	Planting	0.1852	5	1	ns
Notoecta	*Notoncus ectatommoides*	Planting	0.1852	5	1	ns
Sole	*Solenopsis* sp.	Planting	0.1111	3	1	ns
Stigbr	*Stigmacros* brown	Planting	0.1111	3	1	ns
Campeleg	*Camponotus elegans*	Planting	0.0741	2	1	ns
Meloph	*Melophorus* sp.	Planting	0.0741	2	1	ns
Prolyell	*Prolasius* yellow	Planting	0.0741	2	1	ns
Stigbl	*Stigmacros* black	Planting	0.0741	2	1	ns
Tapi	*Tapinoma* sp.	Planting	0.0741	2	1	ns
Campcons	*Camponotus consobrinus*	Planting	0.0370	1	1	ns
Epobo	*Epobostruma* sp.	Planting	0.0370	1	1	ns
Ochet	*Ochetellus* sp.	Planting	0.0370	1	1	ns
Pheioran	*Pheidole* orange	Planting	0.0370	1	1	ns
Poly	*Polyrhachis* sp.	Planting	0.0370	1	1	ns
Merano	*Meranoplus* sp.	Remnant	1.0000	1	0.031	*
Anonsmal	*Anonychomyrma* small	Remnant	0.9502	2	0.07	.
Prolniti	*Prolasius nitidissimus*	Remnant	0.9502	2	0.075	.
Anonbico	*Anonychomyrma biconvexa*	Remnant	0.9194	9	0.031	*
Myrmforf	*Myrmecia forficate*	Remnant	0.8877	4	0.155	ns
Rhytvict	*Rhytidoponera victoriae*	Remnant	0.8060	11	0.091	.
Pheild	*Pheidole* large	Remnant	0.6484	13	0.299	ns
Iridbick	*Iridomyrmex bicknelli*	Remnant	0.5536	21	0.557	ns

Table 5. Indicator values for ant taxa collected in the autumn sampling cycle. Ants counts were square root transformed. Freq is the number of times a species was present among samples (i.e. not its abundance). Significant indicator taxa are highlighted. ns = not significant; * = p<0.05 ; ** p<0.001 .

Ant code	Ant species	Treatment	Indicator Value	Freq	p value	Sig.
Notoecta	*Notoncus ectatommoides*	Planting	0.3937	11	0.119	ns
Monobl	*Monomorium* black	Planting	0.2931	4	0.129	ns
Pheild	*Pheidole* large	Planting	0.2452	6	0.282	ns
Sole	*Solenopsis* sp.	Planting	0.0804	2	0.725	ns
Iridvici	*Iridomyrmex vicina*	Pasture	0.4353	7	0.022	*
Myrmpilo	*Myrmecia pilosula*	Pasture	0.2997	6	0.142	ns
Iriddrom	*Iridomyrmex dromus*	Pasture	0.2903	5	0.138	ns
Pheitasm	*Pheidole tasmaniensis*	Pasture	0.2737	12	0.51	ns
Monol	*Monomorium* large	Pasture	0.2222	2	0.331	ns
Iridbick	*Iridomyrmex bicknelli*	Pasture	0.1481	3	0.541	ns
Poly	*Polyrhachis* sp.	Pasture	0.1111	1	1	ns
Campcons	*Camponotus consobrinus*	Pasture	0.0651	2	1	ns
Amblaust	*Amblyopone australis*	Pasture	0.0556	2	1	ns
Rhytvict	*Rhytidoponera victoriae*	Remnant	0.5008	11	0.041	*
Anonbico	*Anonychomyrma biconvexa*	Remnant	0.4444	4	0.026	*
Monobr	*Monomorium* brown	Remnant	0.4444	4	0.023	*
Rhyttasm	*Rhytidoponera tasmaniensis*	Remnant	0.3478	12	0.215	ns
Prolyell	*Prolasius* yellow	Remnant	0.2222	2	0.309	ns
Myrmforf	*Myrmecia forficate*	Remnant	0.2222	2	0.294	ns
Campeleg	*Camponotus elegans*	Remnant	0.1401	3	0.585	ns
Tapi	*Tapinoma* sp.	Remnant	0.1189	4	0.699	ns
Anonsmal	*Anonychomyrma* small	Remnant	0.1111	1	1	ns
Pheioran	*Pheidole* orange	Remnant	0.1111	1	1	ns
Cera	*Cerapachys* sp.	Remnant	0.1111	1	1	ns
Myrmesur	*Myrmecia esuriens*	Remnant	0.1111	1	1	ns
Stigbl	*Stigmacros* black	Remnant	0.1111	1	1	ns

4.2 Beetles

The collection of beetles recorded in this study was more taxonomically diverse than the ant samples, with 1605 beetles from 22 families and 77 species. A summary of abundance per treatment is given in Table 6, with a complete species list available in Appendix 4.

4.2.1 Beetle abundance

Like the ants, beetles were clearly much more active in the summer than the autumn. The summer trapping cycle collected 1055 beetles from 54 species, with the autumn trapping cycle collecting 550 beetles from 41 species. Beetle abundance was highest overall in the pastures, with 719 or 44.8% of the beetles collected from the 9 pasture sites, 636 or 39.6% collected from the 10 plantation sites, and 250 or 15.6% collected from the 9 remnant patches.

Table 6. Beetle abundance per treatment in each cycle.

Treatment	Summer	Autumn	Overall
Pasture	543	176	719
Plantings	445	191	636
Remnant	67	183	250
TOTAL	**1055**	**550**	**1605**

Fifty of the total 77 beetle species were found in numbers of 5 or less after both trapping cycles, and 30 of these were found as singletons only. The ten most abundant species made up 79.7% of the total numbers.

The most abundant species was the carabid beetle *Aphodius pseudotasmaniae* which made up 37.6% of all beetles collected. This beetle, known as the Common Pasture Cockchafer, was common only in its summer flight season with 600 individuals collected in January and just 4 collected in April. It was also common only in the pasture, with 417 of the total 604 individuals collected from the pasture, 185 from the plantings, and a singleton from the remnant woodlands. The fourth most common beetle, the predatory carabid *Promecoderus ovicollis*, was the only one to be collected from all 3 treatments in both trapping cycles.

The rank abundance curve for beetles (Figure 12) shows a typical profile of invertebrate communities. Few species are common, and many species are uncommon, resulting in a steep curve skewed to the right (ie positive skewed).

Beetle abundance was conducted was found to have a non-parametric distribution (see Appendix 6), and differences in abundance between the three treatments just failed statistical significances ($F_{2,53}$=3.0113, p=0.0577), (Figure 13).

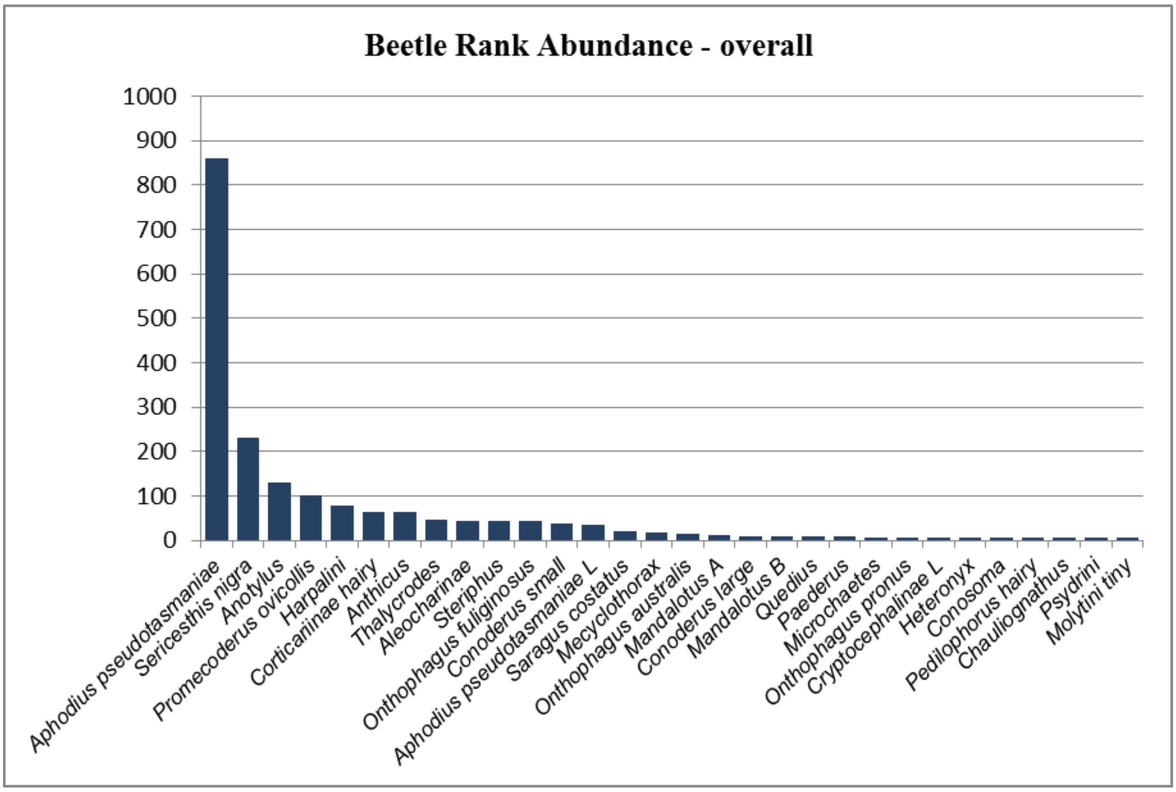

Figure 12. Beetle abundance across all treatments for summer & autumn combined. It was assumed that all traps were in situ for 7 days during the summer cycle and figures for both seasons were then standardized to a 10 day average. Only the most abundant 30 species have been included in the figure. The remaining 47 species were present in total counts of 4 or less.

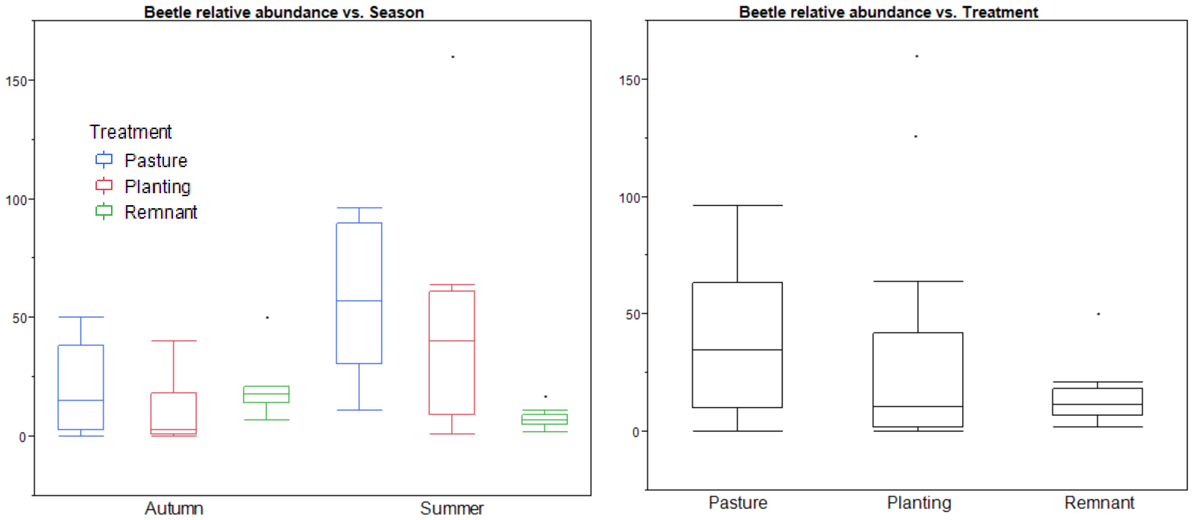

Figure 13. Beetle abundance per treatment in the summer and autumn cycles, and overall abundance per treatment. Values are mean per site within each treatment with a standard error of one. Differences between treatments are not significant ($F_{2,53}$=3.0113, p=0.0577)

4.2.2 Beetle species richness

Beetle species richness was relatively even across the treatments, with 42 species recorded in the pasture sites, 37 species in the plantings, and 41 species in the remnant woodlands for a total of 77 species. A total of 15 species were present in all three treatments. Thirty species were collected as singletons after both trapping cycles. The species richness data was tested for normality and was found to exhibit a normal distribution (see Appendix 6). There was no significant difference in mean species richness between the three treatments ($F_{2,53}$=0.27, p=0.76) (Figure 14).

Despite the remnant woodlands displaying the lowest abundance of beetles, it demonstrated a higher number of unique species, with 23 of the 41 species found in this treatment not being collected from either of the other two treatments. In contrast, the plantings supported only 10 unique species, and the pastures supported 13 unique species. Nine species were found only in the autumn trapping cycle, with 4 species found only in the summer.

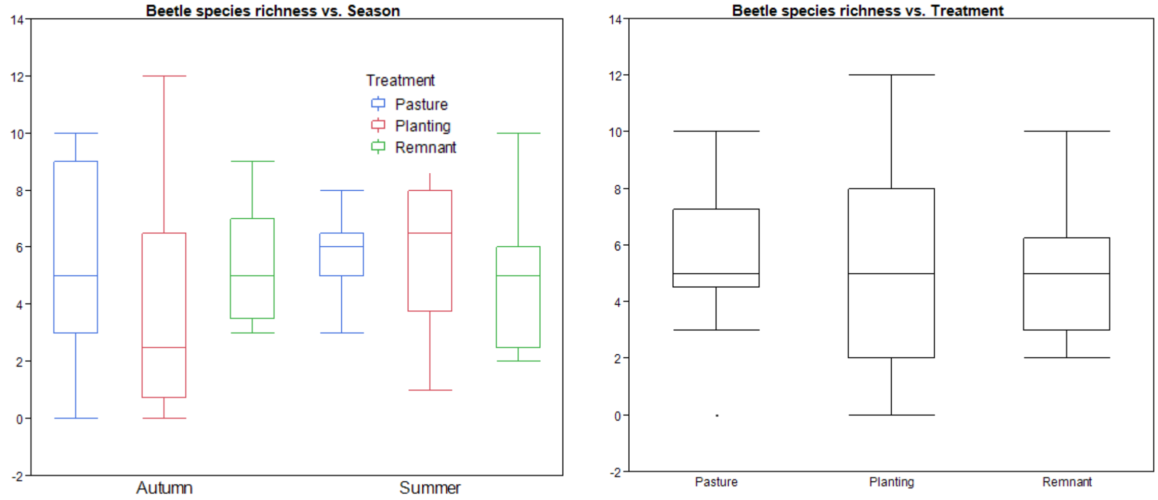

Figure 14. Beetle species richness from both seasons in each treatment. Values are mean species richness per site per treatment with a standard error of one. Differences are not significant ($F_{2,53}$=0.27, p=0.76).

4.2.3 Beetle communities

A non-metric multidimensional scaling (NMDS) ordination of the sites for both seasons combined shows relationships between the beetle communities in each treatment (Figure 15).

The results are similar to that seen in the ant communities, in that the remnant vegetation samples largely occupy ordination space separate from the other habitats, with only a small overlap with the pasture sites. The ant communities of the pasture and plantation sites are somewhat intermingled in the ordination, with the polygon for the planting communities largely overlapping that of the pasture communities.

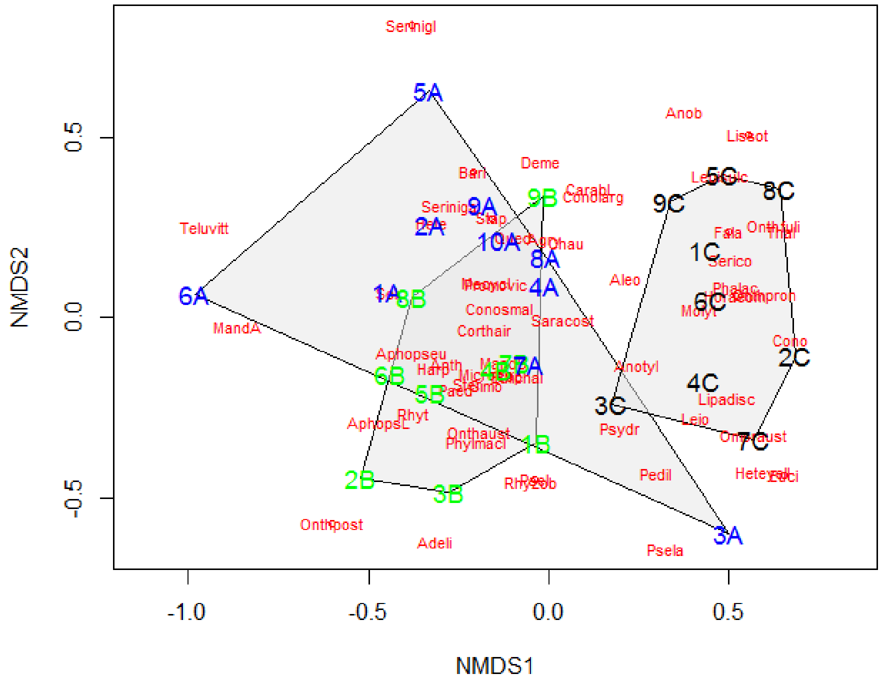

Figure 15. Ordination (NMDS, stress in 2D = 0.140) of sites based upon beetle communities for summer and autumn sample cycles combined. Beetle species are plotted as vectors in the same ordination space.

4.3 Seasonality

The results clearly show that the invertebrate activity differs greatly between the two seasons (Figure 16), and that different species are active at different times of year. Ant abundance was significantly higher in summer compared to autumn ($F_{1,54}$=6.78, p=0.0119). Ant species richness was also significantly higher in the summer ($F_{1,54}$=6.69, p=0.0124). Beetle abundance was significantly higher in the summer than the autumn ($F_{1,54}$=4.22, p=0.049), primarily due to the summer trapping cycle coinciding with the annual dispersal flight of the pasture cockchafer *Aphodius pseudotasmaniae*. There was no significant difference in beetle species richness in the two seasons ($F_{1,54}$=0.95, p=0.3335).

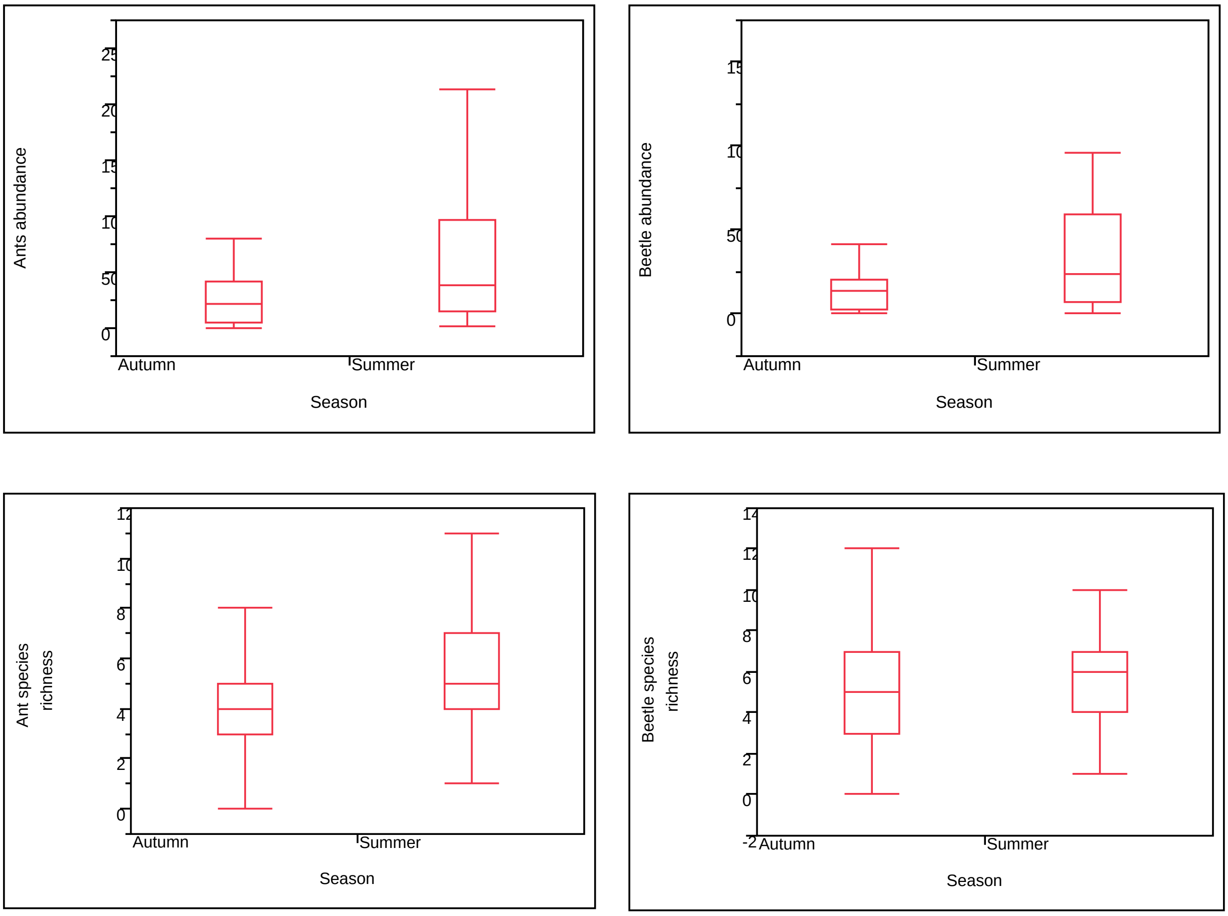

Figure 16. Seasonality differences in abundance and species richness of ants and beetles**.**

Comparing activity levels of ant functional groups discussed in chapter 2.5 is another way of showing differences in the ant communities as the seasons change. The changes in the ant communities related to seasonality can be seen in the two charts in Figure 17. The cold climate specialists and opportunists become more abundant in autumn, whilst activity levels of dominant Dolichoderinae and specialist predators decline considerably. The Hot Climate Specialists were not recorded at all in the cooler trapping cycle.

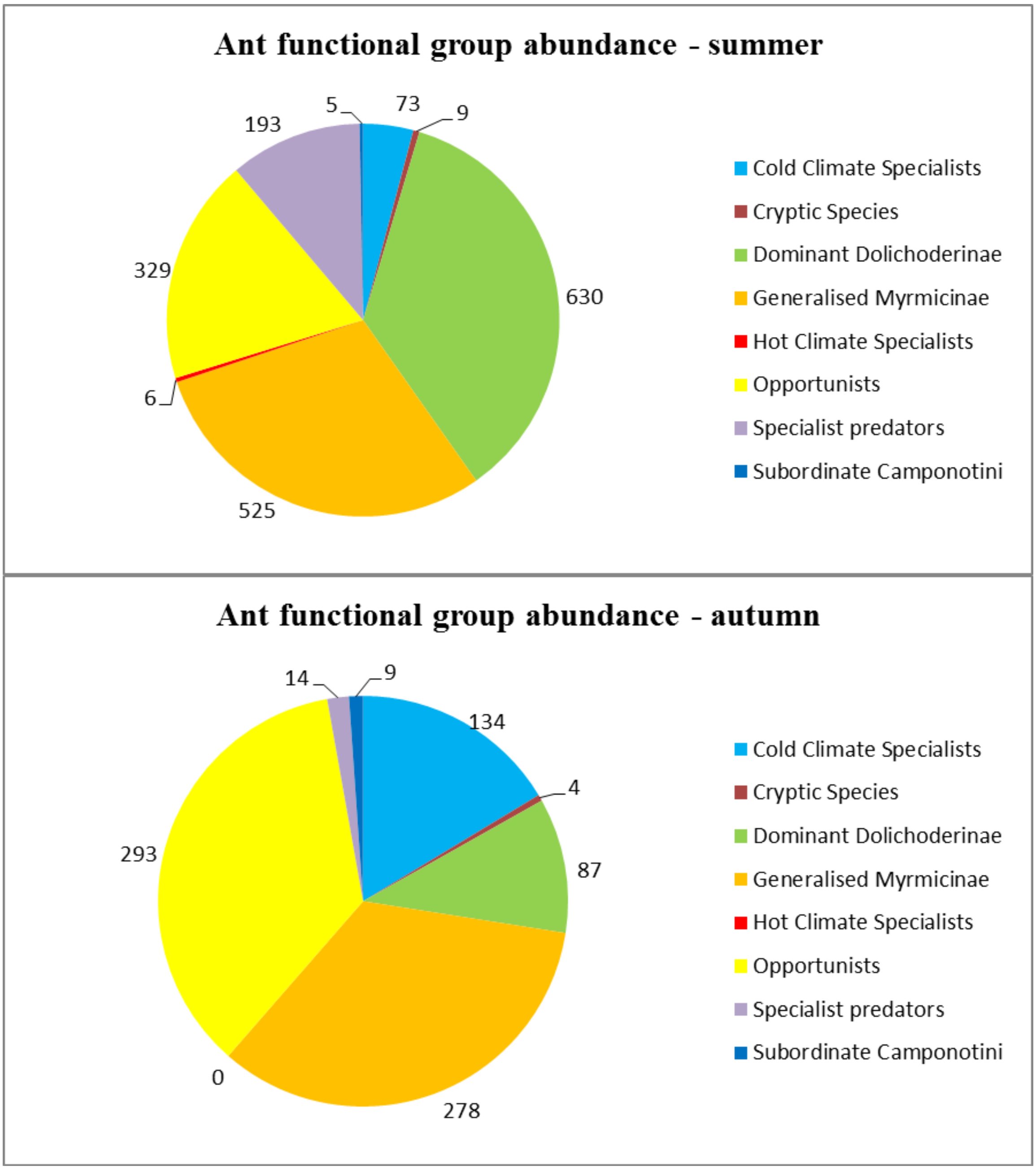

Figure 17. Ant abundance by functional group in summer (top) and autumn (below).

Beetle activity is also shown to be highly variable according to the seasons. An analysis which excluded all beetle species collected as singletons showed that 59.6% of beetle species were recorded in only one of the sampling cycles. The remaining 40.4% which were recorded in both cycles seem to show a clear preference for one season over the other. The 47 beetle species which were collected in numbers of 2 or more are shown graphically with their percentage abundance for each season in Figure 18.

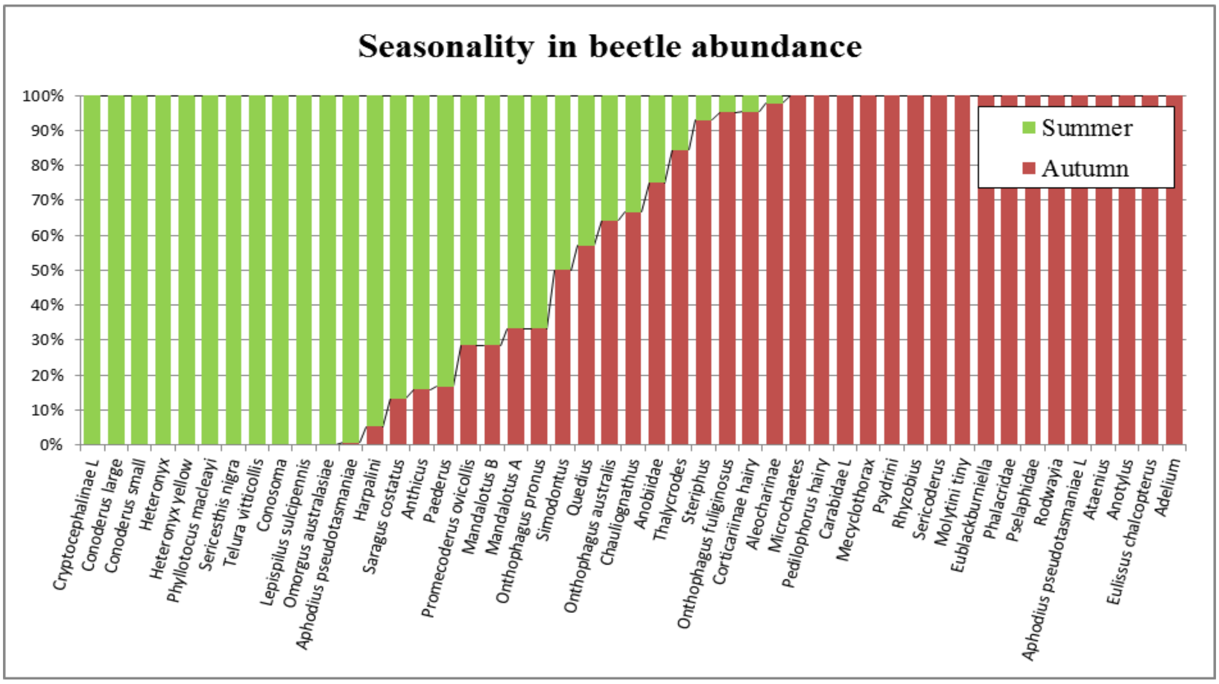

Figure 18. Seasonality in beetle abundance showing individual species' preference for one season or another. Dataset excludes all species collected as singletons.

4.4 Environmental influences

4.4.1 Soil clay content

The percentage of clay in the soil at each site was measured against and ant and beetle abundance and species richness. Soil clay was found to have a significant impact on ant species richness, with the number of ant species declining as the clay content increased ($F_{1,54}$=5.66, p=0.021). The beetle data did not show any significant relationship with soil clay.

4.4.2 Planting area

The area of the each planting was used as an environmental variable to determine whether it influences ant and beetle abundance and species richness. The area of the planting was found only to have a significant impact on beetle abundance, which increased significantly as the area of the planting increased ($F_{1,54}$=4.82, p=0.0415).

4.4.3 Elevation

Elevation was found to have a strongly significant influence on both ant abundance ($F_{1,54}$=10.35, p=0.002) and species richness ($F_{1,54}$=11.52, p=0.001) with both declining as elevation increases. Elevation was not found to affect beetles in any significant way.

4.4.4 Proximity to remnant patch

The distance between the location of the traps and the nearest remnant patch was analysed to test whether the proximity of a remnant patch may be an influencing factor in recolonisation of plantings, as has been seen in minesite rehabilitation (Andersen 1993). No significant relationships were found between distance to a remnant patch and ant abundance, ant species richness, beetle abundance or beetle species richness.

4.4.5 Ground cover

Leaf litter cover and weediness were the only two ground cover variables which showed a significant relationship with invertebrate presence. Ant species richness increased significantly with leaf litter cover ($F_{1,54}$=5.50, p=0.023), whilst beetle abundance decreased significantly ($F_{1,54}$=4.08, p=0.0483).

Ant species richness decreased significantly as weediness increased ($F_{1,54}$=5.30, p=0.025) whilst beetle abundance ($F_{1,54}$=4.87, p=0.032) and species richness ($F_{1,54}$=4.22, p=0.045) both increased with weediness.

The presence of grasses, bracken or shrubs, bare ground, and rocks/moss/lichen were not shown to have any significant relationship with the presence of invertebrates.

4.4.6 Age

Regression analysis undertaken on the plantings shows that ant abundance increases significantly as a planting ages ($F_{1,54}$=5.09, p=0.037). There is no significant relationship between planting age and ant species richness. Beetle abundance ($F_{1,54}$=6.39, p=0.024) and beetle species richness ($F_{1,54}$=4.59, p=0.046) both decline significantly as a planting ages.

A summary of results is shown in Table 7.

Table 7. Summary of analyses of environmental variables against ant abundance, ant species richness, beetle abundance and beetle species richness. ns = not significant, * = p<0.05, ** = p<000.1

Ant abundance	Estimate	DF	R^2	F ratio	p value	Sig.
Soil % Clay	-2.5460	1,54	0.0494	2.8091	0.0995	ns
Planting area	-0.0001	1,54	0.0465	0.8769	0.3614	ns
Elevation	**-0.1329**	1,54	0.1609	10.3540	0.0022	**
Proximity to remnant	0.0072	1,54	0.0245	0.9056	0.3476	ns
Age	**0.1068**	1,54	0.2206	5.0954	0.0367	*
Ground cover - grasses	-0.2381	1,54	0.0209	1.1525	0.0288	ns
Ground cover - leaf litter	0.4607	1,54	0.0503	2.8609	0.0965	ns
Ground cover - bracken/shrubs	-0.2422	1,54	0.0085	0.4646	0.4984	ns
Ground cover - rocks/moss/lichen	0.5437	1,54	0.0102	0.5589	0.4579	ns
Ground cover - bare	0.0643	1,54	0.0005	0.0297	0.8639	ns
Ground cover - weeds	-0.7099	1,54	0.0101	0.5519	0.4607	ns

Ant species richness	Estimate	DF	R^2	F ratio	p value	Sig.
Soil % Clay	**-0.1641**	1,54	0.0948	5.6559	0.0210	*
Planting area	0.0000	1,54	0.0360	0.6727	0.4229	ns
Elevation	**-0.0065**	1,54	0.1758	11.5216	0.0013	**
Proximity to remnant	-0.0001	1,54	0.0171	0.6280	0.4333	ns
Age	0.0142	1,54	0.0100	0.1820	0.6748	ns
Ground cover - grasses	-0.0192	1,54	0.0628	3.6207	0.0624	ns
Ground cover - leaf litter	**0.0291**	1,54	0.0925	5.5027	0.0227	*
Ground cover - bracken/shrubs	-0.0104	1,54	0.0073	0.3966	0.5315	ns
Ground cover - rocks/moss/lichen	0.0469	1,54	0.0352	1.9683	0.1664	ns
Ground cover - bare	0.0155	1,54	0.0148	0.8101	0.3721	ns
Ground cover - weeds		1,54	0.0894	5.3047	0.0251	*

Beetle abundance	Estimate	DF	R^2	F ratio	p value	Sig.
Soil % Clay	0.1661	1,54	0.0005	0.0262	0.8721	ns
Planting area	**0.0002**	1,54	0.2112	4.8203	0.0415	*
Elevation	0.0365	1,54	0.0279	1.5493	0.2186	ns
Proximity to remnant	0.0005	1,54	0.0301	1.1165	0.2977	ns
Age	**-0.2656**	1,54	0.2620	6.3893	0.0241	*
Ground cover - grasses	0.1725	1,54	0.0253	1.3989	0.2421	ns
Ground cover - leaf litter	**-0.3588**	1,54	0.0703	4.0809	0.0483	*
Ground cover - bracken/shrubs	-0.1844	1,54	0.0114	0.6216	0.4339	ns
Ground cover - rocks/moss/lichen	-0.6747	1,54	0.0363	2.0347	0.1595	ns
Ground cover - bare	0.3329	1,54	0.0340	1.8980	0.1740	ns
Ground cover - weeds	**1.3380**	1,54	0.0827	4.8703	0.0316	*

Beetle species richness	Estimate	DF	R^2	F ratio	p value	Sig.
Soil % Clay	-0.1411	1,54	0.0533	3.0414	0.0869	ns
Planting area	0.0001	1,54	0.1610	3.4530	0.0796	ns
Elevation	0.0039	1,54	0.0477	2.7058	0.1058	ns
Proximity to remnant	-0.0005	1,54	0.0390	1.4604	0.2347	ns
Age	**-0.0038**	1,54	0.2032	4.5916	0.0460	*
Ground cover - grasses	0.0020	1,54	0.0005	1.1410	0.2902	ns
Ground cover - leaf litter	-0.0089	1,54	0.0067	0.3621	0.5499	ns
Ground cover - bracken/shrubs	-0.0230	1,54	0.0270	1.4961	0.2266	ns
Ground cover - rocks/moss/lichen	0.0081	1,54	0.0008	0.0435	0.8356	ns
Ground cover - bare	0.0175	1,54	0.0144	0.7880	0.3786	ns
Ground cover - weeds	**0.1014**	1,54	0.0725	4.2194	0.0448	*

Chapter 5 Discussion and Conclusion

The primary aim of this study was to survey the invertebrate activity in restoration plantings and determine whether there is a significant difference to the invertebrate activity recorded in pasture and remnant woodland patch controls. The secondary aim was to identify which environmental variables, if any, influence the recolonisation of a restoration planting by invertebrates. Environmental factors considered include soil clay content, the size of the planting, elevation, proximity to a remnant woodland patch, and ground cover and weediness.

Whilst the suitability of invertebrates as monitoring tools in restoration projects is widely accepted, their use is best indicated in medium-long term studies (Underwood & Fischer 2006). In practice, their application in this short term study of just 3 months has resulted in a dataset which shows clear patterns of succession, but a longer-term study would likely produce results with stronger statistical significance. There are a number of factors which have influenced this outcome, including flaws in the research design, limitations on the available sites, and resource limitations.

When considering potential environmental influences, it became difficult to discount or disprove the influence of environmental factors with certainty due to high variation in the samples and low number of replicates. However, some strong links with invertebrate activity were detected, which are discussed further in 5.3.

5.1 Summary of top 5 ant species by abundance

Table 8 lists the five most abundant ant species collected in this study.

Table 8. Top five ant species by rank abundance

Rank	Species	Summer	Autumn	TOTAL
1	*Pheidole tasmaniensis*	339	128	467
2	*Rhytidoponera victoriae*	144	171	315
3	*Iridomyrmex bicknelli*	286	3	289
4	*Rhytidoponera tasmaniensis*	131	110	241
5	*Myrmecia pilosula*	187	8	195

Landscape restoration programs cause high levels of disturbance in a landscape, caused by the requisite soil tilling and preparation, the practice of planting seedlings, and ongoing monitoring by landowners and plant scientists. The relative abundance of ant species is strongly influenced by habitat disturbance (Andersen 1991) and disturbed areas often show a particularly high abundance of opportunistic species such as *Rhytidoponera* spp., which made up two of the top 5 species by abundance in this study. The top 5 species are listed in Table 8, and there is nothing remarkable about these particular species making up the top 5 by abundance, for reasons which follow. Overall ant abundance was highest in the plantings, despite the plantings not having the highest species richness. This supports the results of Cunningham *et al* (2005) who demonstrated that eucalypt plantings supported fewer, but more abundant, insect species when compared to remnant eucalypt forest.

The most abundant species, *P. tasmaniensis*, was most common in the plantings with a mean of 12.25 individuals per site in the plantings, mean concentrations of 9.67 per site in the pastures, and 2.56 in the remnant patches. This reflects the description of genus *Pheidole* given by Andersen (1991) which states they are commonly found in disturbed areas.

The ectatommine *Rhytidoponera* is a common genus with over 100 species in Australia with *R. tasmaniensis* and *R. victoriae* both important seed dispersers in Tasmania. Both can be extremely abundant species, and commonly occur together (Andersen 1991). Both species were much more abundant in the remnant woodland sites than they were in the pastures or plantations, with a mean of 6.78 and 14.67 ants per site respectively.

Genus *Iridomyrmex* is a group of highly active and aggressive ants with 79 described Australian species, most of which are scavengers (Shattuck 1999) and commonly inhabit open sunny habitats (Andersen 1991). In this study, *I. bicknelli* was most common in the plantings with a mean of 11.3 individuals per site, compared with a mean of 2.5 individuals per site in the pastures, and 0.9 individuals per site in the remnant woodland patches.

Myrmecia pilosula, commonly known as the “Jack Jumper” ant, is the most common of the species in its genus in Tasmania and is equally abundant in woodlands and open spaces

(Andersen 1991). In this study it was predominantly collected in the plantings, with one site alone (Site 6A) recording 161 individuals out of the total count of 195 in just the summer cycle. Whilst due care was taken not to set traps near obvious nesting places, this highly skewed result could be caused by the traps being set close to either the foraging trails or the nests of this species. This type of skewed result could be minimised if, rather than actual abundance counts, a scale of abundance was used (Andersen 1993), such as 1 = 1 ant, 2 = 2-5 ants, 3 = 6-20 ants, 4 = 21-50 ants and 5 = >50 ants. Whilst the ordinations used data which had been square-root transformed, repeating the analyses shown in Figures 8 and 9 using a scale of abundance could result in some interesting outcomes.

5.2 Summary of top 5 beetle species by abundance

Table 9 lists the five most abundant beetle species collected in this study.

Table 9. Top five beetle species by rank abundance

Rank	Species/morphospecies		Summer	Autumn	TOTAL
1	Scarabaeidae	*Aphodius pseudotasmaniae*	600	4	604
2	Scarabaeidae	*Sericesthis nigra*	163	0	163
3	Staphylinidae	*Anotylus* sp	0	132	132
4	Carabidae	*Promecoderus ovicollis*	55	22	77
5	Latridiidae	*Corticariinae* hairy	3	61	64

The dominance of a single native pest species, the pasture cockchafer or blackheaded cockchafer *Aphodius pseudotasmaniae,* causes a highly skewed result in beetle abundance. It must be noted that this species was almost absent from the autumn sampling, suggesting the summer trapping cycle may have coincided with its annual mating flight. Another cockchafer beetle, *Sericesthis nigra*, was also highly abundant in summer and completely absent in autumn. Infestations by cockchafer beetles are common in pastures, and have the potential to cause major economic loss through damage to sown pasture grasses, weed invasion of open spaces, and the cost of chemical control (Allen 1977, Pauley & Miller 1993).

As discussed in 5.1, using a scale of abundance rather than actual abundance counts may help minimise any distortion caused by very high counts of a small number of pest species.

Members of the staphylinid genus *Anotylus* are predatory beetles which feed on dipteran eggs and larvae (Good & Giller 1988) and are important natural enemies of many insect pests in the agroecosystem (Good & Giller 1988). Another member of this group of predatory carabid beetles is *Promecoderus ovicollis* which is known to be common in sheep pastures in the Midlands of Tasmania (McQuillan 1997).

The family Latridiidae, commonly known as minute brown scavenger beetles, includes 29 genera which occupy nearly all major terrestrial habitats. The food preference of most latridiid beetles is unknown, however they have been observed in decaying vegetation and stored food products (Lord *et al* 2010).

5.3 The influence of environmental variables

The analyses of environmental variables and their influence on invertebrate presence provided some unexpected outcomes in this study. The strongest correlation for all of the environmental variables came with elevation. As elevation increased, ant abundance and species richness both decreased significantly. Elevation is likely to be a surrogate for climatic conditions such as temperature and precipitation. Although this result is strongly supported by the literature (Sanders *et al* 2003; Botes *et al* 2005), the strength of the significance is noteworthy. Across the sites there was a moderate range in elevation from 102m to 580m which did not result in any strong vegetation gradients, and none of the sites extended to a sub-alpine environment. In contrast, beetle activity was not impacted by elevation.

I was not able to demonstrate that proximity to a remnant patch is an important factor in recolonisation of a planting, despite to literature showing it has an influence over ant species richness (Andersen 1993). Similarly, I did not find any significant relationship between the area of a planting and ant activity even though remnant patches greater than 3 hectares are thought to have particular importance for fauna conservation (Lindenmayer & Hobbs 2004). This is despite 4 of the 10 plantings in this study exceeding 3 hectares in area, with the average planting area being 8.13 ha. To properly assess the influence of this particular environmental variable, it would be useful to repeat the study using more plantings, classifying the plantings into size categories for analysis.

One measurement not fully explored in this study was the complexity of the vegetation in the habitat. Ant species richness is known to generally increase as plant species richness increases (Majer 1977, Boulton *et al* 2005) and this could be one factor in why the remnants patches demonstrate the highest ant species richness. However, as the analysis only considered % ground cover, rather than a calculated biomass assessment, it failed to identify any relationship between habitat complexity and invertebrate activity.

Certain components of the ground cover were found to have an influence on invertebrate activity. Leaf litter was positively correlated with ant species richness and negatively correlated with beetle abundance. The increase in ant species richness is supported by findings by Majer (1985) and Lassau & Hochuli (2004). It would be of value to explore this further by measuring leaf litter by volume at each site, rather than as percentage coverage. In practice, the inclusion of deciduous trees in a restoration planting can facilitate the development of a leaf litter ecosystem that can be sustained throughout the year (Jansen 2008). The decline in beetle abundance has no relationship with ant species richness, with a bivariate comparison of the two variables showing that on average, beetle abundance will actually increase with ant species richness ($F_{1,54}$=10.98, p=0.0016) as it does with ant abundance ($F_{1,54}$=1944.95, p<0.0001) (see Appendix 11).

The results showing the relationship between the weediness of a site and invertebrate activity were unexpected. Ant species richness declined significantly as weediness increased ($F_{1,54}$=5.31, p=0.025) whilst there was an increase in beetle abundance ($F_{1,54}$=4.87, p=0.032) and beetle species richness ($F_{1,54}$=4.22 p=0.045). Few previous studies have examined the relationship between weediness and ant activity, other than one which investigated which of 4 particular weeds a fire ant native to Mexico, *Solenopsis geminata*, prefers to eat (Risch & Carroll 1986). It’s possible that weediness is a surrogate for other factors not measured in this study, such as soil mineral content, or soil moisture. However, the result suggests that effective weed management in plantings may aid the return of ant species, whilst concurrently managing beetle pest species. I am unfamiliar with weed management practices in agriculture, and whilst this is beyond the scope of this study, the application of herbicides to manage weeds in a landscape would also impact on the invertebrate communities (Pereira *et al* 2005). Further research may uncover ways to manage weeds organically on a large scale, without negatively impacting invertebrates.

Further analysis shows that as a planting ages, ant abundance increases significantly, while beetle abundance and beetle species richness decrease. Andersen (1995) found that the rate of change in insect activity stalls once a restoration reaches 8 years of age, however this study did not have enough replicates of each age group to adequately test this theory. The beetle results are likely attributable to the fact that beetles were found to be far more common in a pasture habitat, so as a planting grows to resemble less of a pasture, it follows that beetle activity might decline. It is important to note that the methods used in this study would have captured ground dwelling beetles only. A comprehensive survey of beetle communities should include multiple trapping methods such as light traps, sweep netting and sticky traps. What this data doesn't confirm is causality, given that environmental factors such as ground cover and canopy cover are likely to change as a planting ages, so the results may not simply be attributable to the age of the trees at a site. Regardless, the data suggesting that ant abundance improves with time is encouraging.

5.4 Improvements in site selection

In this study, I attempted to test if ground dwelling invertebrate communities change over time, by looking at a chronosequence of restoration plantings aged from 2 years to 25 years. Unfortunately there were too many other environmental variables influencing the results, and not enough replicates to enable a confident assessment of how the ant communities change over time. To more accurately analyse temporal biotic changes in a restoration planting, an ideal research design would involve repeated sampling of the same sites over many years. This kind of longitudinal study would involve time and resources which unfortunately were not available for my research.

In this study, ten representative restoration plantings were chosen by a Greening Australia representative out of a possible 38 restoration planting sites in Tasmania. The plantings displayed varying ages, elevations, soil types, and distances to remnant patches. They were also widely spread spatially, with the distance between planting 1A and 9A, for example, being approximately 76km. Whilst there is merit in choosing sites at random to ensure an unbiased research design, unfortunately this may allow too much environmental variation to be introduced to the study. The result is that a comparison of the invertebrate communities in two particular plantings was not a simple comparison of like with like. This made it more difficult to compare the plantings as a group with the two sets of control sites.

Additionally, due to the relatively low number of replicates, it was difficult in to pin point the key environmental drivers that may affect the ant communities in a particular planting. If the project was to be repeated, a better design might involve fewer sites that are closer together spatially, and more replicate sites of the same age. This would reduce variation from geographical features such as altitude, climate and soil type, should enable a better analysis of the small scale variables which are easier for a landowner to control, such as ground cover, planting age, and proximity to a remnant woodland patch.

For future research, I would also recommend setting more traps at each site (ideally between 5 and 10, rather than just 3) in order to have more replicates for each site, leading to better site-level estimates. This would better capture variation within a site or within a treatment, not just variation between treatments. I would also recommend choosing restoration plantings that are closer spatially in order to reduce natural variation that would occur from geographical distance. Finally, having a number of sites at specific ages, for example 4 sites aged 3 years, 4 sites aged 12 years, and 4 sites aged 25 years, may enable more solid data regarding the changes attributable to the age of the planting.

5.5 Limitations in pitfall trapping methods

A review of invertebrate sampling techniques by Neville & Yen (2007) considered the values and limitations of pitfall trapping compared to other sampling methods including direct searching, flight intercept traps, sweep netting, sticky traps, vacuum sampling, and nine other methods. They found a trend towards the use of multiple methods; in the early 1990s, few researchers used methods other than pitfall traps, but in the late nineties most researchers employed multiple techniques. Pitfall trapping was the most commonly used technique for testing the conservation significance of natural ecosystems, however in an agricultural landscape, multiple techniques are recommended. Due to financial and temporal restrictions, and the focus of this study being ground-dwelling ants, only pitfall traps were used. It must be noted that this has the potential to discount ants in flight, particularly for the summer cycle, and also many beetle species which would be better sampled using other techniques.

As my study used pitfall trapping exclusively as a sampling method, this has resulted in a number of limitations which could potentially influence the outcomes and results:

a) Pitfall traps are biased towards ground-dwelling invertebrates, meaning that most winged invertebrates, or at least their winged adult stages, will be under-represented. As no other sampling techniques were utilised in this project, there is expected to be a bias towards ground-dwelling invertebrates in the results. By focusing on ants in the results of the study, it is expected that bias will be minimised.

b) Each pitfall trap was covered with a 10cm x 10cm piece of 12mm steel wire mesh. This modification may have prevented some larger species of beetles and spiders from becoming subjects of the study.

c) The disturbance created by setting a pitfall trap may increase the activity of ants for a period of up to a week after establishment and for up to 2 weeks for carabid beetles (Greenslade 1973). These so called “digging-in effects” can be reduced by setting the traps and leaving covered or closed for a period of a week, before being opened and made active. Due to financial and time constraints, no allowance was made to reduce the potential “digging-in effects” and traps were left opened from the moment they were set. As a consequence, the results may show elevated numbers of ant and beetle activity. However, it is unlikely that this would affect relative abundance. The autumn cycle is less likely to be influenced by digging-in effects as the trap design meant the outer cup was left in place in the ground after the summer cycle, removing the need to dig and disturb the soil again in autumn. Regardless, any digging-in effect is distributed equally across all the samples in the experiments and unlikely to bias the results for any particular treatment or location in comparative analyses.

There is no standard method used commonly amongst researchers when laying pitfall traps (Neville & Yen 2007) but efforts were made to mitigate the limitations imposed by the exclusive use of pitfall traps in this study. Firstly, the spacing between a series of pitfall traps can significantly influence results, with the ideal separation between traps being at least 5m to ensure best representation of species present in the local environment (Ward *et al* 2001). Placing the traps with closer or wider spacings has no significant effect on the number of individuals caught in each trap (Ward *et al* 2001).

Trap diameter is a potential source of variation in the pitfall trapping technique. Various receptacles can be used for pitfall traps, from test tubes to buckets, which raises the question of which trap diameter is most effective. Traps with diameters of 18mm, 42mm, 86mm and 135mm diameters were compared with regards to species richness by Abensperg-Traun & Steven (1995) in Western Australia. Whilst the two largest traps were no more efficient than traps with a 42mm diameter, only the 86mm and 135mm traps caught invertebrates greater than 10mm in length. The 135mm trap was not found to be any more effective than the 86mm trap. The smaller 18mm trap caught significantly fewer invertebrate species. From these results, it could be suggested that a trap of around 86mm diameter presents the most efficient size with the least disturbance to the environment. Following these recommendations, the trap used in this study was a plastic cup of 90mm in diameter.

Research studies vary in the dimensions of traps used, the preservative chosen, the number of traps used per site, and the number of days the traps are set for. Researchers also employ a variety of traps, including glass test tubes, plastic cups, plastic ice cream containers, plastic take-away containers, or even a 20L bucket. It could be difficult to compare different projects which have used differing pitfall trapping techniques. It is more important to ensure consistency in trapping methods and techniques for every trap within a particular study to ensure there is no bias in the results. However, truly effective bio-monitoring should be undertaken with a range of sampling methods, and if this study was to be repeated with greater resources at hand, that is certainly my recommendation.

5.6 Temporal restrictions

Two sampling cycles were undertaken – one in summer and one in autumn. To obtain a more comprehensive representation of the invertebrate communities in a landscape across a calendar year, it would be ideal to add a late winter or spring sampling cycle also. This is because of the seasonality seen in the activity of many invertebrates. However, given the time restrictions on completing this research project, it was not possible to include a spring sampling cycle in the results. If resources allowed, the repetition of all summer, autumn and springtime trapping cycles for another year or more would dilute the effects of periods of heavy rain or flooding, drought, or extreme temperatures which may influence the sampling efficacy. Development of a species accumulation curve following multiple trapping cycles might help to reveal when all species expected to occupy a site have been captured.

5.7 Unknown variables

The presence and stocking intensity of sheep in the pastures affects invertebrate abundance level, with ant and scarabid beetle numbers likely to increase with a higher concentration of sheep per hectare (Hutchinson and King 1980). Other invertebrates may be less abundant with a higher intensity of livestock (Hutchinson and King 1980). Having knowledge of the stocking rate in the control pasture sites might help explain anomalies in the dataset in some of the control sites.

The possible use of agrochemicals at the sites is another unknown but potentially influential variable. With the exception of Site 5A "School Block," all plantings sites were adjacent to pasture with sown exotic grasses, and weeds were observed at all pasture sites and in 3 of the planting sites. It is unknown whether any landowners have used fertilisers to promote growth of crops or pasture grasses, herbicides to manage weeds, or pesticides to manage insect pests. If so, this is likely to impact the results for the pasture sites due to the effects of both the agrochemical (Pereira *et al* 2005) and the heavy machinery used to apply it (Cerdà & Jergenson 2008).

5.8 Summary of conclusions

This research project was conducted as a pilot study to test whether invertebrate succession in restoration plantings could be detected using a particular protocol. Whilst it has failed to document significant changes in invertebrate populations in the plantings compared with adjacent pasture at any significant level, a clear progression in ant abundance and species richness was observed. A clear decline in beetle numbers from the pasture to the planting was evident, whilst beetle species richness remained steady across all treatments. The study has gathered a sound set of baseline data which will be useful in further monitoring programs at the planting sites involved.

Ordinations suggest a strong overlap in the ant communities in the pastures and the plantings for both seasons, whilst the community in the remnants is mostly separate. This suggests that the main source of the ants recruited to the plantings appears to be the pastures, not the nearby woody remnants as originally hypothesised. The failure to detect any significant relationship between distance to remnant patch and the ant numbers supports this finding.

Indicator values for 4 ant species may be helpful in ongoing monitoring, to determine when the invertebrate complexity of a restoration planting has reached a level similar to that of nearby remnants – suggesting successful ecological rehabilitation. The four indicator species identified in this study should be important components of any ongoing monitoring of these restoration plantings. If, as the plantings mature, the ant communities in the plantings become more similar to the remnant patches, then it is likely that three species which are currently indicative of the remnant patches will no longer indicative of that habitat type. This would clearly show that the plantings are no longer distinctly different to the remnant patches in terms of their habitat values.

The analyses of environmental variables serve to demonstrate that a landowner wishing to increase ant activity in a restoration planting, or on the land in general, should favour land management practices that preserve leaf litter on a site, and if possible manage weeds organically without the use of agrochemicals.

Further sampling in the short term to build on this data would create a more complete dataset representing invertebrate species from across the calendar year. It would also allow additional analysis of environmental factors to better understand their influence on invertebrate colonisation in restoration plantings.

In summary, this study has shown that restoration plantings can enable a landowner to increase invertebrate activity on the land, regardless of the size of the planting, or its proximity to a remnant patch. This is valuable knowledge for any landowner who might be reluctant or unable financially to dedicate large tracts of land to restoration, or who may not have land available near a healthy remnant. Invertebrates respond better to plantings at lower elevations and with lower clay content, but the factors which a land owner can control are leaf litter cover, weed management, and time, remembering that the age of the planting is more important than where it is located or how much grass is underfoot.

I hope the results of my research will encourage land managers in rural environments to earmark parts of their land to restoration planting, with the aim of improving the agricultural matrix, improving habitat for invertebrates, and creating landscape connectivity for birds and mammals.

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APPENDICES

Appendix 1

Coordinates of all sites, including restoration plantings, pasture controls and remnant woodland patch controls.

Site No.	Name	Longitude	Latitude	Elevation (m)
1A	Meadowbank	146.8214	-42.6389	310
2A	Olympic Landcare	146.7723	-42.5406	113
3A	Uralla	146.8493	-42.5462	115
4A	Dungrove	146.8872	-42.2688	552
5A	School Block	147.0170	-42.3869	367
6A	North Stockman	147.1973	-42.4965	239
7A	Grassy Hut	147.0803	-42.3985	436
8A	Sorrell Springs	147.4595	-42.2422	380
9A	Woodland Park	147.6139	-42.2795	396
10A	Curringa	146.7780	-42.5638	119
1B	1B Pasture	146.8198	-42.6395	306
2B	2B Pasture	146.7782	-42.5630	126
3B	3B Pasture	146.8483	-42.5466	112
4B	4B Pasture	146.8863	-42.2690	544
5B	5B Pasture	147.0115	-42.4029	382
6B	6B Pasture	147.2003	-42.4949	252
7B	7B Pasture	147.0796	-42.3981	437
8B	8B Pasture	147.4037	-42.2880	496
9B	9B Pasture	147.4062	-42.2935	490
1C	1C Remnant	146.8180	-42.6382	319
2C	2C Remnant	146.7728	-42.5724	102
3C	3C Remnant	146.8505	-42.5458	142
4C	4C Remnant	146.8937	-42.2713	580
5C	5C Remnant	147.0211	-42.4020	443
6C	6C Remnant	147.0483	-42.3882	424
7C	7C Remnant	147.0372	-42.3883	433
8C	8C Remnant	147.4034	-42.2867	502
9C	9C Remnant	147.4070	-42.2935	498

Appendix 2

Environmental features of the restoration plantings

Site	1A	2A	3A	4A	5A
Property or Site Name	Meadowbank	Olympic Landcare	Uralla	Dungrove	School Block
Latitude	146.8214	146.7723	146.8493	146.8872	147.017
Longitude	-42.6389	-42.5406	-42.5462	-42.2688	-42.3869
Year established	2011	2000	1998	2011	1994
Age at sampling	2	13	15	2	19
Area (ha)	17.8	0.6	2.7	31.1	1.6
Elevation (m)	310	113	115	552	367
Monocalyptus present/absent	present	present	present	present	present
Symphiomyrtus present/absent	absent	absent	present	absent	present
Vegetation type / Dominant Eucalypt	*Eucalyptus pauciflora* *Eucalyptus tenuiramis*	*E. pauciflora* *Acacia verticillata*	*Eucalyptus globulus* *Eucalyptus delegatensis* *E. pauciflora*	*E. pauciflora*	*Eucalyptus amygdalina* *Eucalyptus viminalis* *E. pauciflora*
Understory - Dominant species	*Acacia dealbata*	*Bursaria spinosa* *Banksia marginata* *A. dealbata* *Acacia melanoxylon*	*Dodonaea viscosa* *A. dealbata*	*None*	*None*
Ground cover type - Dominant species	Scotch thistle, *Pteridium esculentum*, introduced grasses	Mixed grass species	Mixed grass species, moss, stones	Mixed grasses, *Lomandra* spp, exotic grasses	Thin cover of leaf litter
% Grasses	0	50	50	30	20
% Leaf litter	0	30	20	0	75
% Bracken / shrubs	15	0	0	0	0
% rocks / moss / lichen	0	2	10	10	0
% bare	80	20	20	50	5
% weeds	5	0	0	10	0
Scotch thistle	present	absent	absent	absent	absent
Parent Geology	Sandstone	Sandstone	Sandstone	Sandstone	Sandstone
Soil % clay	23	29	15	19	29
Distance to remnant patch (m)	200	3100	80	300	950

Appendix 2 cont.

Site	6A	7A	8A	9A	10A
Property or Site Name	North Stockman	Grassy Hut	Sorrell Springs	Woodland Park	Curringa
Latitude	147.1973	147.0803	147.4595	147.6139	146.778
Longitude	-42.4965	-42.3985	-42.2422	-42.2795	-42.5638
Year established	1988	2011	2003	2003	1993
Age at sampling	25	2	10	10	20
Area (ha)	4.5	17	2.8	2.7	0.5
Elevation (m)	239	436	380	396	119
Monocalyptus present/absent	present	present	present	present	absent
Symphiomyrtus present/absenst	absent	absent	absent	absent	absent
Vegetation type / Dominant Eucalypt	*E. pauciflora* *E. tenuiramis* *Acacia dealbata*	*E. pauciflora* *E. tenuiramis*	*E. pauciflora*	*E. pauciflora* *Pinus radiata*	*Dodonaea viscosa* *Acacia melanoxylon* *Eucalyptus leucoxylon*
Understory - Dominant species	*B. spinosa* *B. marginata*	*A. dealbata*	*A. dealbata*	*D. viscosa* *B. spinosa*	*A. dealbata* *A. melanoxylon* *D. viscosa* *E. leucoxylon*
Ground cover type - Dominant species	Highly grazed exotic grasses	Mixed grass species, moss, stones	native and exotic grasses	Exotic grasses	*Epacris* spp
% Grasses	75	70	30	40	20
% Leaf litter	20	0	20	50	49
% Bracken / shrubs	0	0	0	0	0
% rocks / moss / lichen	0	0	0	0	2
% bare	5	20	50	10	29
% weeds	0	10	0	0	0
Scotch thistle	absent	present	absent	absent	absent
Parent Geology	Sandstone	Mudstone	Sandstone	Basalt	Dolerite
Soil % clay	22	24	22	24	23
Distance to remnant patch (m)	1750	660	1000	1960	1000

Appendix 3

Complete list of ant species and morphospecies collected in both sampling cycles, by rank abundance, with abundance counts for each treatment and cycle. All species are members of the Order Formicidae.

Species / Morphospecies	**Summer**			**Autumn**			**TOTAL**
	pasture	**planting**	**remnant**	**pasture**	**planting**	**remnant**	
Pheidole tasmaniensis	148	189	2	26	58	44	467
Rhytidoponera victoriae	10	26	108	10	5	156	315
Iridomyrmex bicknelli	46	224	16	0	2	1	289
Rhytidoponera tasmaniensis	18	60	53	35	6	69	241
Myrmecia pilosula	4	174	9	1	6	1	195
Iridomyrmex vicina	68	92	12	0	13	2	187
Pheidole large dark	67	49	41	15	14	0	186
Anonychomyrma biconvexa	1	0	118	0	0	48	167
Notoncus ectatommoides	11	13	0	67	48	2	141
Tapinoma	0	8	45	0	5	7	65
Monomorium brown large	0	0	0	35	29	0	64
Iridomyrmex dromus	25	25	0	5	8	0	63
Prolasius yellow	0	0	29	0	0	16	45
Monomorium large	0	0	0	0	43	0	43
Monomorium brown	1	0	11	0	0	10	22
Pheidole orange	0	0	17	0	0	4	21
Anonychomyrma small	0	0	3	0	0	8	11
Stigmacros black	0	0	9	0	0	1	10
Myrmecia forficata	0	2	3	0	0	4	9
Solenopsis	5	0	2	1	1	0	9
Stigmacros brown	0	0	8	0	0	0	8
Camponotus elegans	0	1	1	0	2	3	7
Camponotus consobrinus	0	2	0	0	2	1	5
Amblyopone australis	0	0	2	0	1	1	4
Meranoplus	0	0	4	0	0	0	4
Prolasius nitidissimus	0	0	3	0	0	0	3
Polyrhachis	0	0	1	0	1	0	2
Melophorus	1	0	1	0	0	0	2
Cerapachys	0	0	0	0	0	1	1
Myrmecia esuriens	0	0	0	0	0	1	1
Epobostruma	0	0	1	0	0	0	1
Ochetellus	0	1	0	0	0	0	1

Appendix 4

Complete list of beetle species and morphospecies collected in both sampling cycles, by rank abundance, with abundance counts for each treatment and cycle.

Family	**Species / Morphospecies**	**Summer**			**Autumn**			**TOTAL**
		Pasture	**Planting**	**Remnant**	**Pasture**	**Planting**	**Remnant**	
Scarabaeidae	*Aphodius pseudotasmaniae*	415	184	1	2	2		604
Scarabaeidae	*Sericesthis nigra*	45	117	1				163
Staphylinidae	*Anotylus*				2	82	48	132
Carabidae	*Promecoderus ovicollis*	7	38	10	15	4	3	77
Latridiidae	*Corticariinae* hairy	1	1	1	25	36		64
Carabidae	*Harpalini*	26	27		2	1		56
Nitidulidae	*Thalycrodes*			7			38	45
Staphylinidae	*Aleocharinae*			1	11	16	17	45
Curculionidae	*Steriphus*	3			26	14		43
Scarabaeidae	*Onthophagus fuliginosus*			2			40	42
Anthicidae	*Anthicus*	23	9			6		38
Scarabaeidae	*Aphodius pseudotasmaniae* L				35			35
Elateridae	*Conoderus* small	1	22	3				26
Carabidae	*Mecyclothorax*				9	6	3	18
Anobiidae	*Anobiidae*		2	2	12			16
Tenebrionidae	*Saragus costatus*	1	4	8		2		15
Scarabaeidae	*Onthophagus australis*	3		2	8		1	14
Curculionidae	*Mandalotus* A	2	4		3			9
Byrrhidae	*Microchaetes*				1	7		8
Byrrhidae	*Pedilophorus* hairy				2	2	3	7
Curculionidae	*Mandalotus* B		5		2			7
Elateridae	*Conoderus* large	1	4	2				7
Staphylinidae	*Quedius*		3			4		7
Cantharidae	*Chauliognathus*		2		1	1	2	6
Carabidae	*Psydrini*						6	6
Curculionidae	*Molytini* tiny						6	6
Scarabaeidae	*Onthophagus pronus*			4			2	6
Staphylinidae	*Paederus*	5				1		6
Chrysomelidae	*Cryptocephalinae* L		5					5
Leiodidae	*Eublackburniella*				1	3	1	5
Scarabaeidae	*Ataenius*				5			5
Scarabaeidae	*Heteronyx*		5					5
Staphylinidae	*Conosoma*			5				5
Tenebrionidae	*Adelium*				5			5
Scarabaeidae	*Heteronyx* yellow		1	3				4
Scarabaeidae	*Telura vitticollis*		4					4
Tenebrionidae	*Lepispilus sulcipennis*			4				4
Carabidae	*Carabidae* L				1	1	1	3
Coccinellidae	*Rhyzobius*				1		2	3
Corylophidae	*Sericoderus*						3	3
Pselaphidae	*Pselaphidae*				2	1		3
Staphylinidae	*Eulissus chalcopterus*				1	2		3
Carabidae	*Simodontus*	1			1			2
Phalacridae	*Phalacridae*						2	2
Ptiliidae	*Rodwayia*						2	2
Scarabaeidae	*Phyllotocus macleayi*	1	1					2
Trogidae	*Omorgus australasiae*			2				2
Carabidae	*Clivina*	1						1
Carabidae	*Demetrida*	1						1
Carabidae	*Homethes*						1	1
Carabidae	*Rhytisternus*	1						1
Chrysomelidae	*Calomela*			1				1

Appendix 4 cont.

Chrysomelidae	*Cryptocephalus*		1					1
Chrysomelidae	*Eboo*			1				1
Chrysomelidae	*Monolepta*		1					1
Curculionidae	*Amycterinae*		1					1
Curculionidae	*Baris*		1					1
Curculionidae	*Listronotus bonariensis*	1						1
Curculionidae	*Merimnetes australis*			1				1
Curculionidae	*Naupactus*				1			1
Curculionidae	*Poropterus*			1				1
Elateridae	*Agrypnus*		1					1
Eucinetidae	*Eucinetus*			1				1
Leiodidae	*Leiodidae*			1				1
Lucanidae	*Lissotes*			1				1
Pselaphidae	*Pselaphini*				1			1
Scarabaeidae	*Liparetrus discipennis*			1				1
Scarabaeidae	*Onthophagus intro*	1						1
Scarabaeidae	*Onthophagus posticus*				1			1
Scarabaeidae	*Scitala* L						1	1
Scarabaeidae	*Sericesthis nigrolineata*		1					1
Scydmaenidae	*Horaeomorphus*						1	1
Staphylinidae	*Falagria*			1				1
Staphylinidae	*Pselaphinae*	1						1
Staphylinidae	*Staphylininae*		1					1
Tenebrionidae	*Isopteron*	1						1
Tenebrionidae	*Seirotrana*	1						1

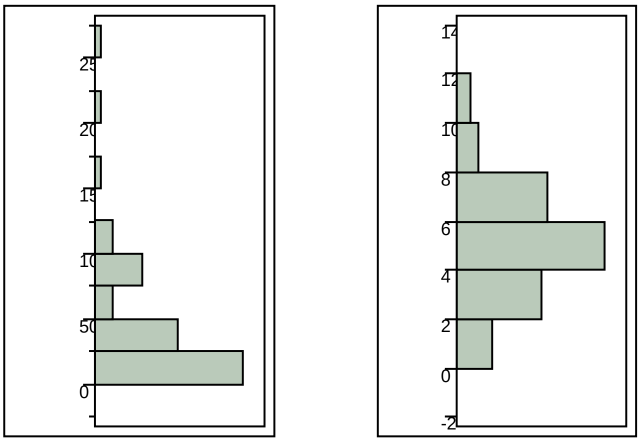
Appendix 5

Complete list of spider species and morphospecies collected in both sampling cycles, by rank abundance, with abundance counts for each treatment and cycle.

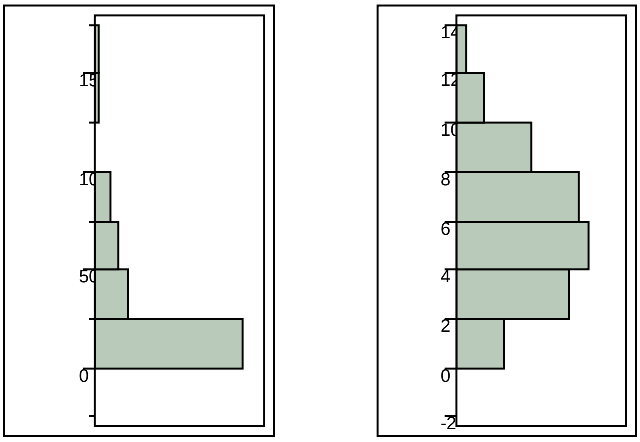
Family	Species/ morphospecies	Summer			Autumn			TOTAL
		Pasture	Planting	Remnant	Pasture	Planting	Remnant	
Lycosidae	*Artoria*	52	30	12	22	21	20	157
Lycosidae	*Lycosidae* imm	35	17	6	59	13	16	146
Linyphiidae	*Linyphiidae bifida*	0	0	0	104	0	0	104
Linyphiidae	*Linyphiidae*	7	1	0	17	19	7	51
Zodariidae	*Zodariidae*	2	4	9	13	1	13	42
Gnaphosidae	silver grey	6	8	5	3	1	1	24
Lycosidae	*Lycosa*	18	1	1	0	0	1	21
Gnaphosidae	*Gnaphosid* pale tiny	0	0	0	5	2	13	20
Gnaphosidae	*Gnaphosidae* dark	4	2	5	2	1	2	16
Zoridae	*Zoridae*	1	4	8	0	0	3	16
Amaurobiidae	*Amaurobiidae*	0	0	0	2	2	11	15
Lycosidae	*Venatrix*	1	1	0	4	6	1	13
Dysderidae	*Dysdera crocata*	1	7	0	1	1	0	10
Corinnidae	*Supunna*	0	7	1	0	0	1	9
Prodidomidae	*Myandra bicincta*	2	6	1	0	0	0	9
Thomisidae	*Cymbacha* cf	2	3	1	0	2	1	9
Lamponidae	*Lampona* black	3	1	0	1	0	1	6
Lycosidae	*Lycosa godeffroyi*	1	5	0	0	0	0	6
Oonopidae	*Oonopidae*	0	1	3	0	1	1	6
Salticidae	*Salticidae*	0	1	3	0	0	1	5
spider	pink abd black spot	2	3	0	0	0	0	5
Theridiidae	*Theridiidae* A	1	0	0	3	1	0	5
Zoridae	orange hindlegs	0	0	0	0	0	4	4
Lamponidae	banded legs	0	2	1	0	0	0	3
Salticidae	dark legs	0	0	2	0	1	0	3
Desidae	*Badumna*	0	1	1	0	0	0	2
Desidae	*Teatta*	0	0	2	0	0	0	2
Araneidae	*Agryodes*	0	0	1	0	0	0	1
Clubionidae	*Clubiona*	0	0	1	0	0	0	1
Hahnidae	*Hahnidae*	0	0	0	0	0	1	1
Lamponidae	*Lamponidae*	0	0	1	0	0	0	1
Lycosidae	*Venatrix* spotted	0	0	0	0	0	1	1
Nicodamidae	*Nicodamidae*	0	0	0	0	0	1	1
Theridiidae	*Theridiidae* B	0	0	1	0	0	0	1
Thomisidae	*Sidymella*	0	1	0	0	0	0	1
Thomisidae	*Stephanopis*	0	0	1	0	0	0	1

Appendix 6

Histograms showing pattern of distribution for ant abundance, ant species richness, beetle abundance and beetle species richness.



Distribution histograms showing non-parametric distribution for ant abundance (L) and normal distribution for ant species richness (R).



Distribution histograms showing non-parametric distribution for ant abundance (L) and normal distribution for ant species richness (R).

Appendix 7

Soil chemistry data for all sites, sourced from ASRIS – the Atlas of Australian Soils.

http://www.**asris**.csiro.au/

Site	Treatment	Soil Type (ASRIS)	% clay (ASRIS)	% N	% C
1A	Planting	loam / silty loam / sandy clay loam	23	N/A	N/A
2A	Planting	loam / silty loam / sandy clay loam	29	N/A	N/A
3A	Planting	Sandy loam	15	0.375	4.18
4A	Planting	Sandy loam	19	0.487	3.84
5A	Planting	loam / silty loam / sandy clay loam	29	0.265	3.36
6A	Planting	loam / silty loam / sandy clay loam	22	0.180	1.79
7A	Planting	loam / silty loam / sandy clay loam	24	0.583	6.39
8A	Planting	loam / silty loam / sandy clay loam	22	0.230	2.45
9A	Planting	loam / silty loam / sandy clay loam	24	0.390	5.05
10A	Planting	loam / silty loam / sandy clay loam	23	N/A	N/A
1B	Pasture	loam / silty loam / sandy clay loam	23	N/A	N/A
2B	Pasture	loam / silty loam / sandy clay loam	23	N/A	N/A
3B	Pasture	Sandy loam	15	0.230	2.06
4B	Pasture	Sandy loam	19	0.233	2.63
5B	Pasture	loam / silty loam / sandy clay loam	29	0.490	5.19
6B	Pasture	loam / silty loam / sandy clay loam	22	0.133	3.38
7B	Pasture	loam / silty loam / sandy clay loam	22	0.520	5.78
8B	Pasture	Clay loam	30	N/A	N/A
9B	Pasture	Clay loam	30	N/A	N/A
1C	Remnant	loam / silty loam / sandy clay loam	23	N/A	N/A
2C	Remnant	loam / silty loam / sandy clay loam	23	N/A	N/A
3C	Remnant	Sandy loam	15	0.350	4.62
4C	Remnant	Sandy loam	19	0.260	5.29
5C	Remnant	loam / silty loam / sandy clay loam	22	0.317	6.21
6C	Remnant	loam / silty loam / sandy clay loam	22	N/A	N/A
7C	Remnant	loam / silty loam / sandy clay loam	22	1.215	5.19
8C	Remnant	Clay loam	30	0.377	6.49
9C	Remnant	Clay loam	30	0.377	6.49

Appendix 8

Ant species and their assignment to functional groups, following Andersen (1995).

Cold Climate Specialists
Monomorium brown
Monomorium brown large
Monomorium large
Notoncus ectatommoides
Prolasius nitidissimus
Prolasius yellow
Stigmacros black
Stigmacros brown

Cryptic Species
Amblyopone australis
Solenopsis sp.

Dominant Dolichoderinae
Anonychomyrma biconvexa
Anonychomyrma small
Iridomyrmex bicknelli
Iridomyrmex dromus
Iridomyrmex vicina

Generalised Myrmicinae
Pheidole large dark
Pheidole orange
Pheidole tasmaniensis

Hot Climate Specialists
Melophorus sp.
Meranoplus sp.
Ochetellus sp.

Opportunists
Rhytidoponera tasmaniensis
Rhytidoponera victoriae
Tapinoma sp.

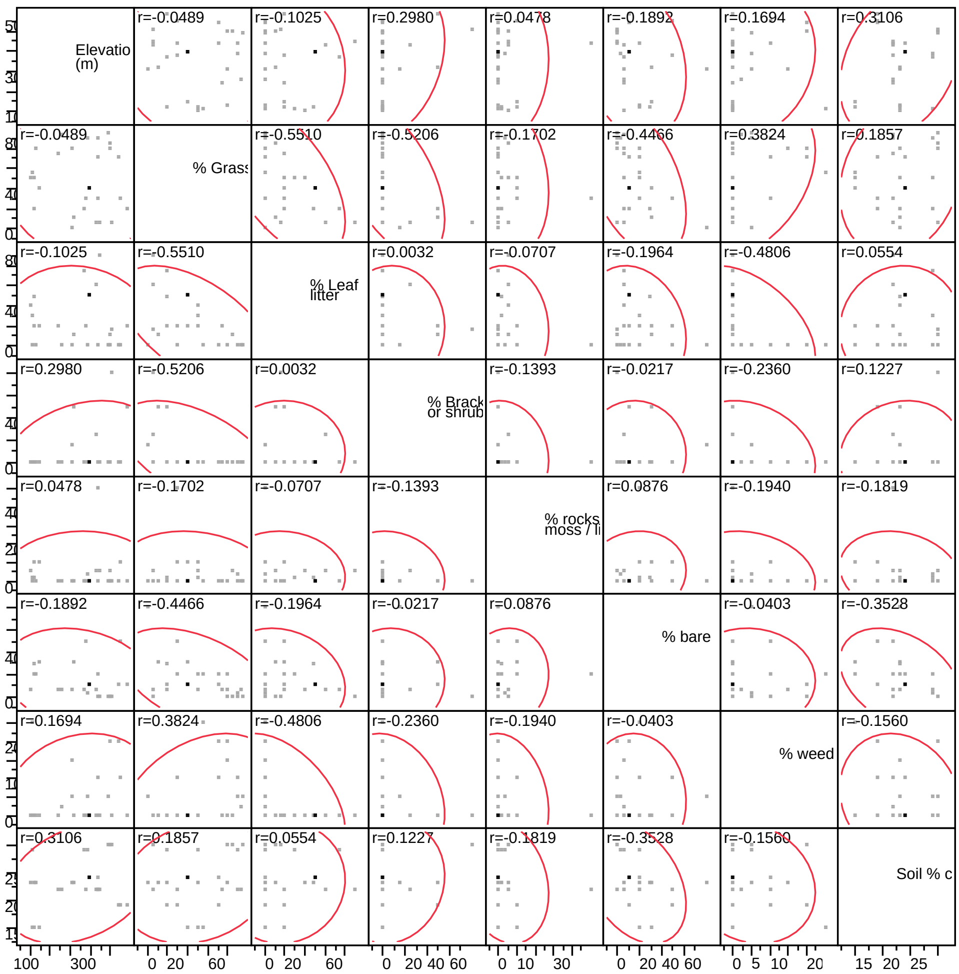
Specialist predators
Cerapachys sp.
Epobostruma sp.
Myrmecia esuriens
Myrmecia forficata
Myrmecia pilosula

Subordinate Camponotini
Camponotus consobrinus
Camponotus elegans
Polyrhachis sp.

Appendix 9

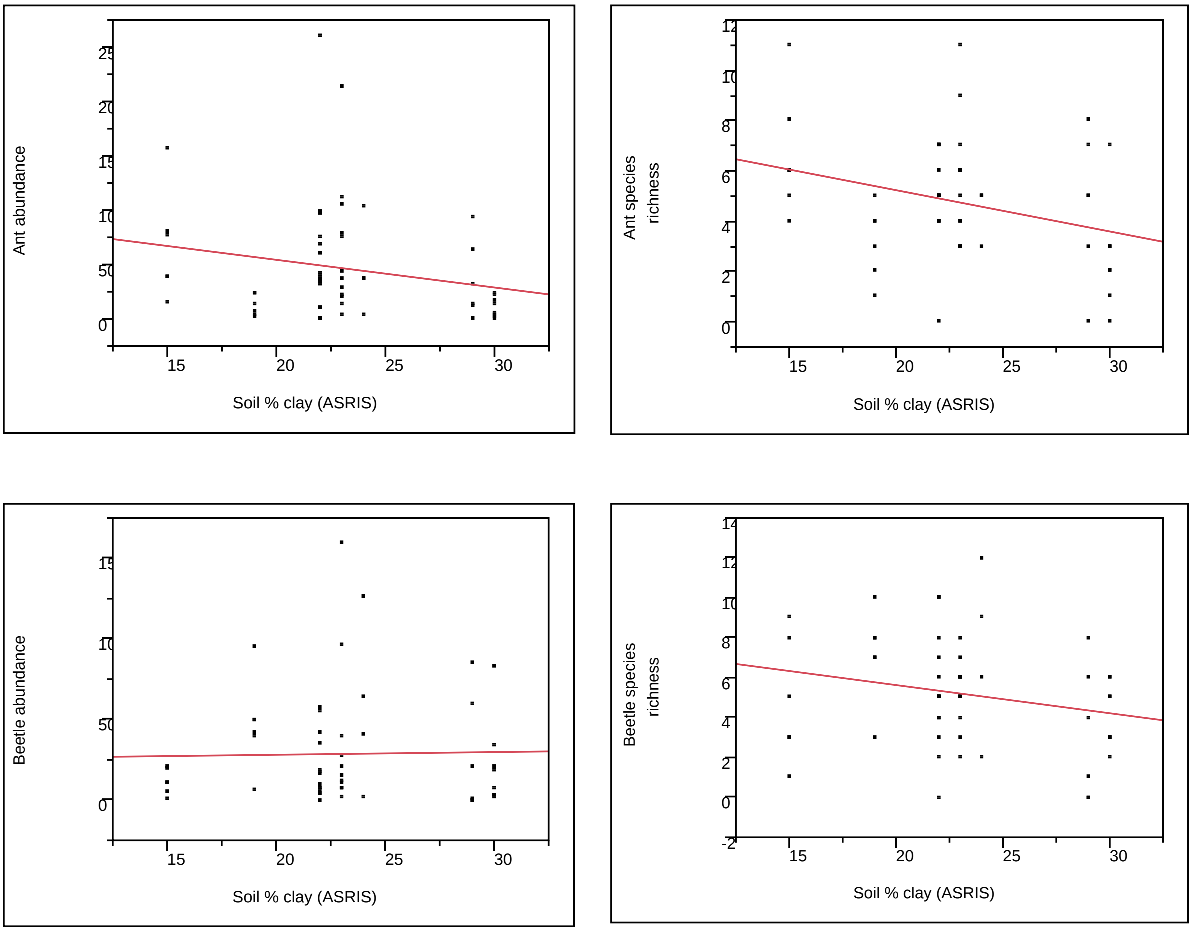
Multivariate Correlation of ground cover variables

	Elevation	Grasses	Leaf litter	Bracken / shrubs	Rocks / moss / lichen	Bare	Weeds
Elevation (m)	1						
Grasses	-0.0489	1					
Leaf litter	-0.1025	-0.5510	1				
Bracken or shrubs	0.2980	-0.5206	0.0032	1			
Rocks / moss / lichen	0.0478	-0.1702	-0.0707	-0.1393	1		
Bare	-0.1892	-0.4466	-0.1964	-0.0217	0.0876	1	
Weeds	0.1694	0.3824	-0.4806	-0.2360	-0.1940	-0.0403	1
Soil % clay	0.3106	0.1857	0.0554	0.1227	-0.1819	-0.3528	-0.1560



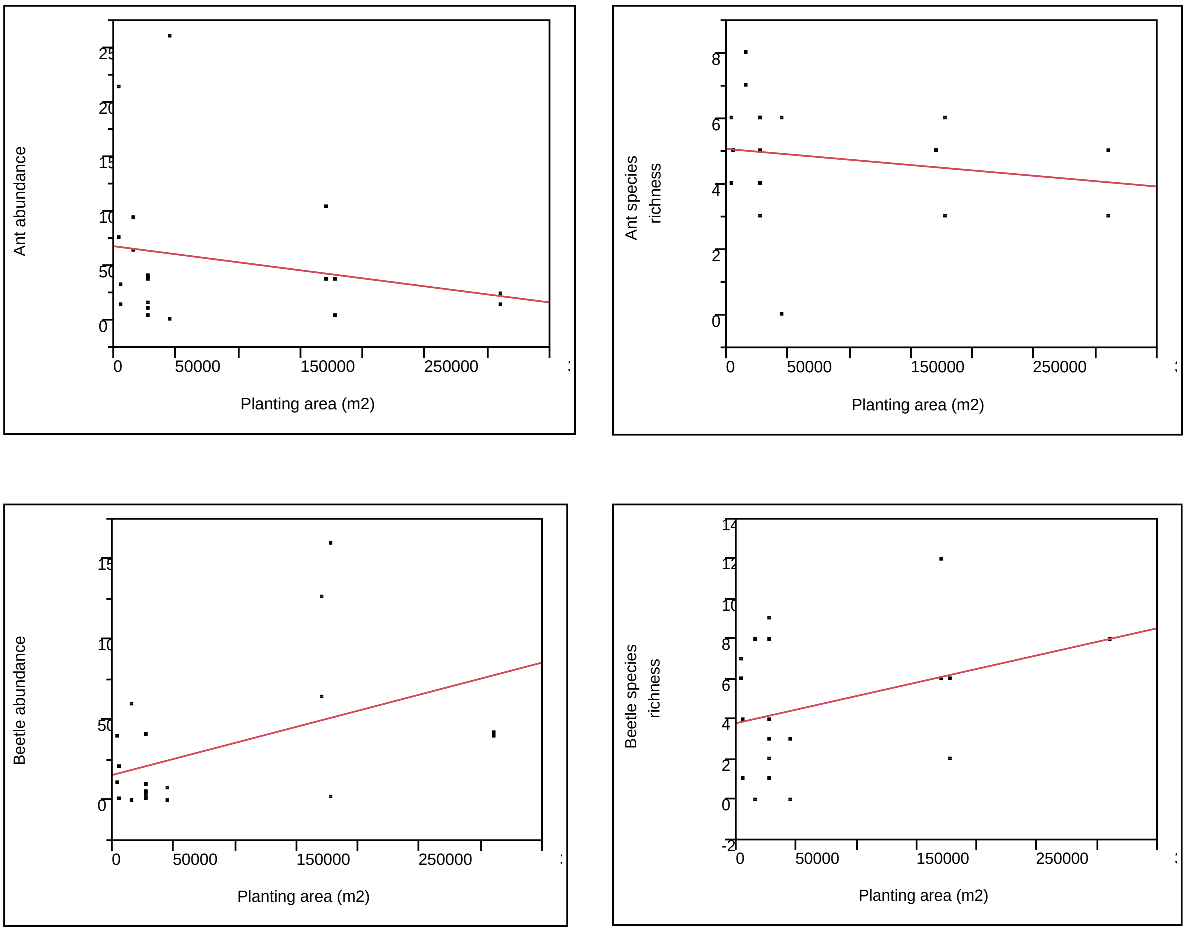
Appendix 10

Environmental variables and their correlation with ant and beetle abundance and species richness.



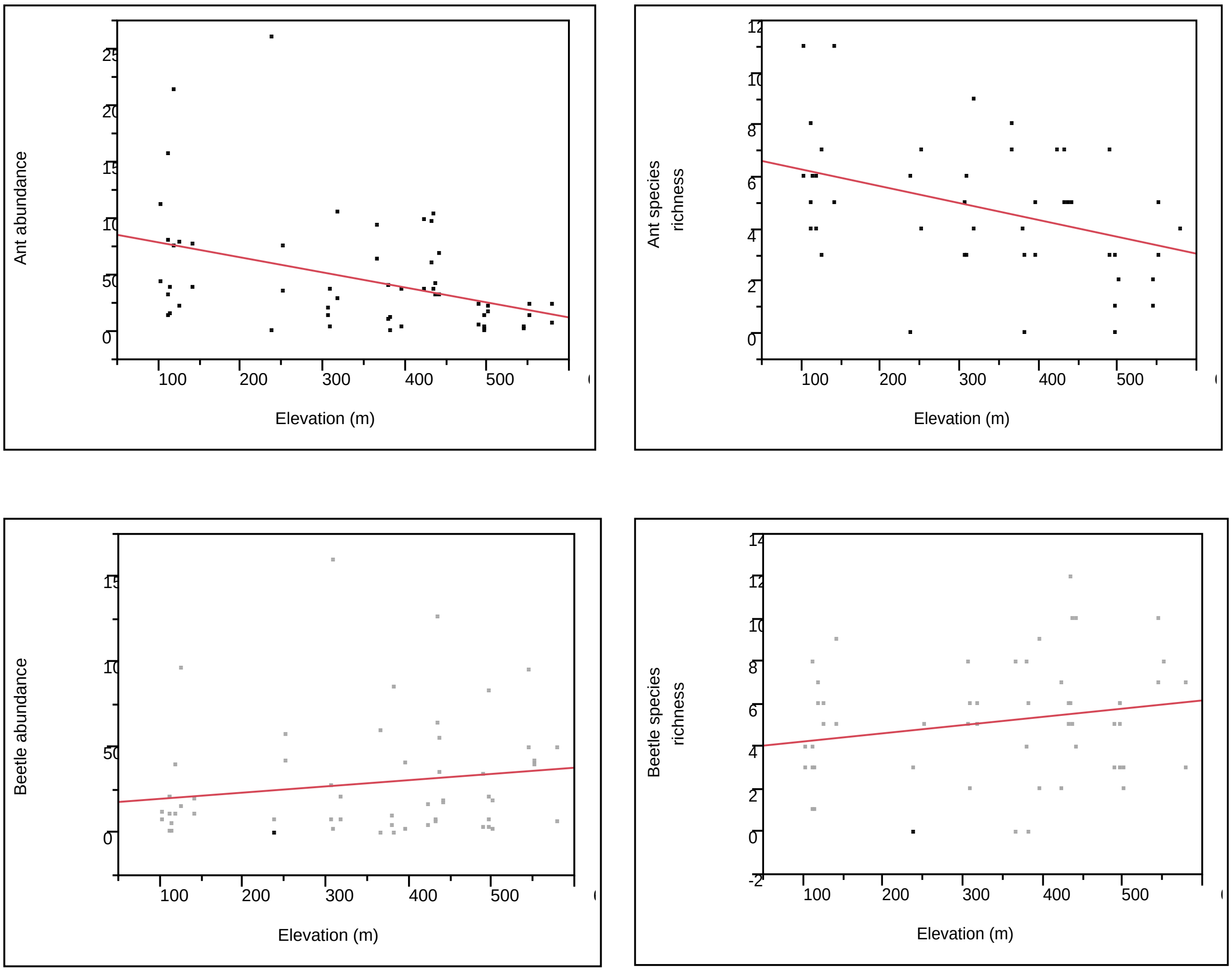
Regression showing relationship between soil clay content and ant and beetle abundance and species richness. Analysis conducted on data from all sites. Relationship with ant species richness is significant ($F_{1,54}$=5.66, p=0.0210).

Appendix 10 cont.



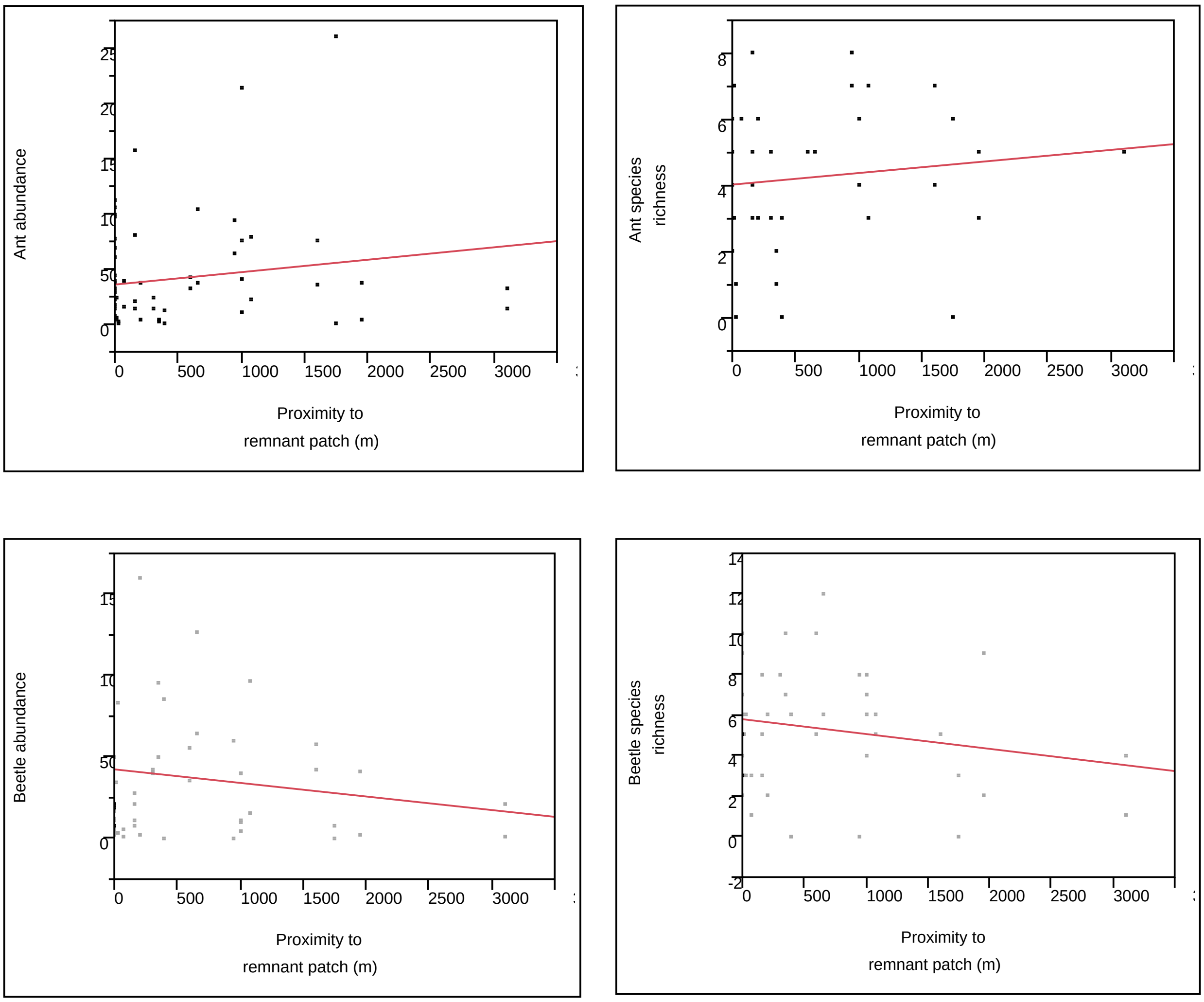
Regression showing relationship between area of planting and ant and beetle abundance and species richness. Analysis conducted on data from planting sites only. None of the relationships are significant.

Appendix 10 cont.



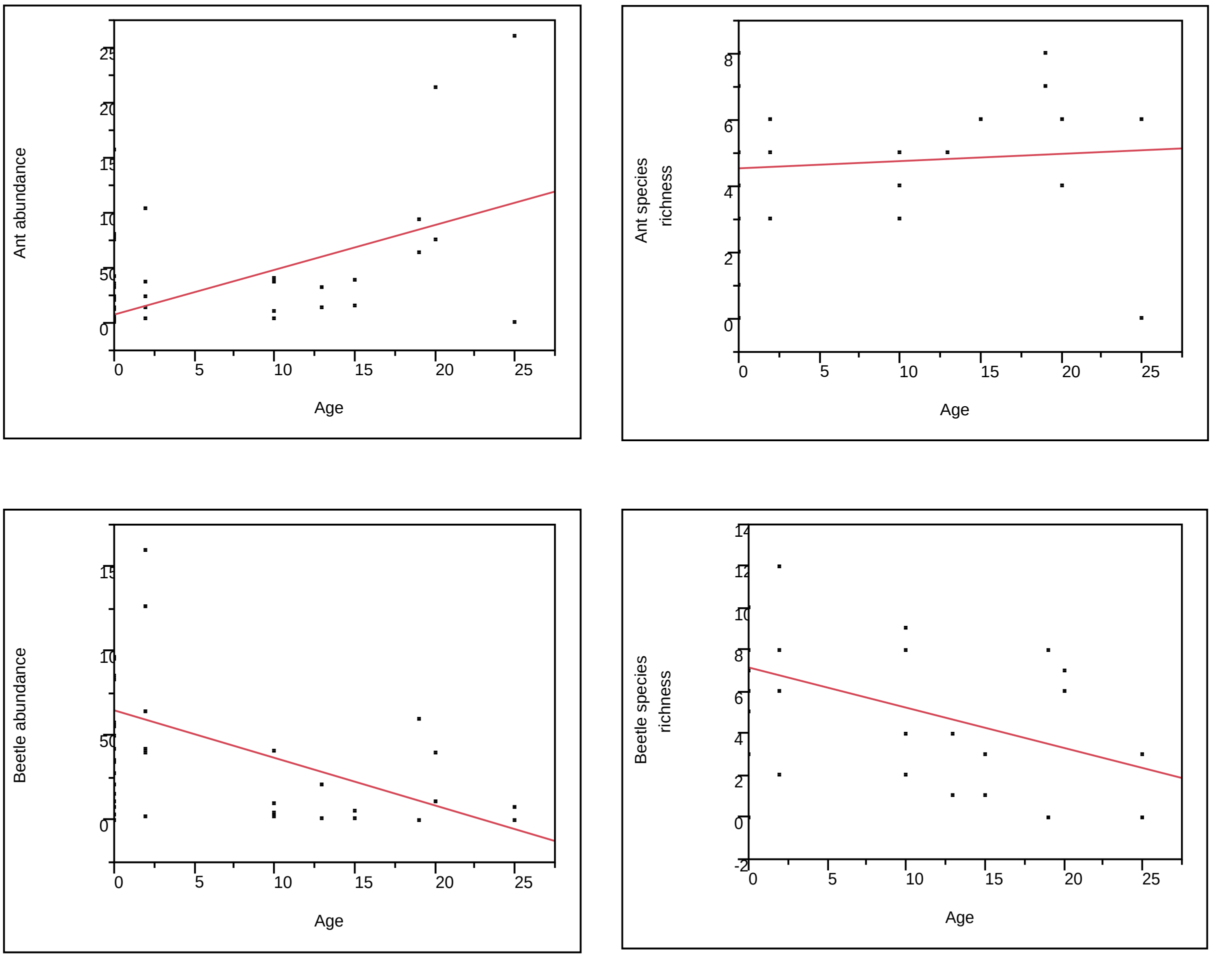
Regression showing relationship between elevation and ant and beetle abundance and species richness. Analysis conducted on data from all sites. Relationship is significant for ant abundance ($F_{1,54}$=10.35, p=0.0022) and ant species richness ($F_{1,54}$=11.52, p=0.0013).

Appendix 10 cont.



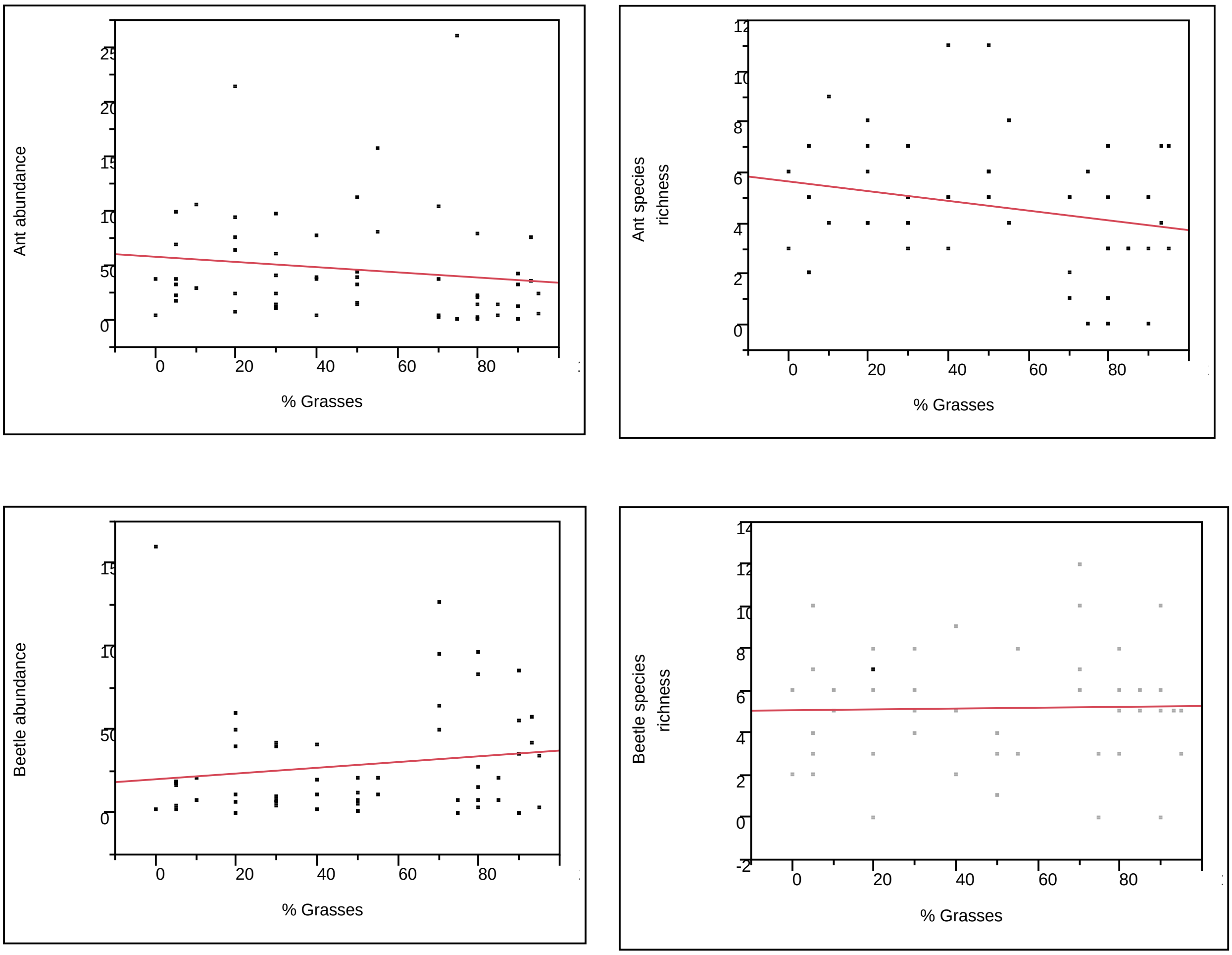
Regression showing relationship between proximity to a remnant patch and ant and beetle abundance and species richness. Analysis conducted on data from planting sites only. None of the relationships are significant.

Appendix 10 cont.



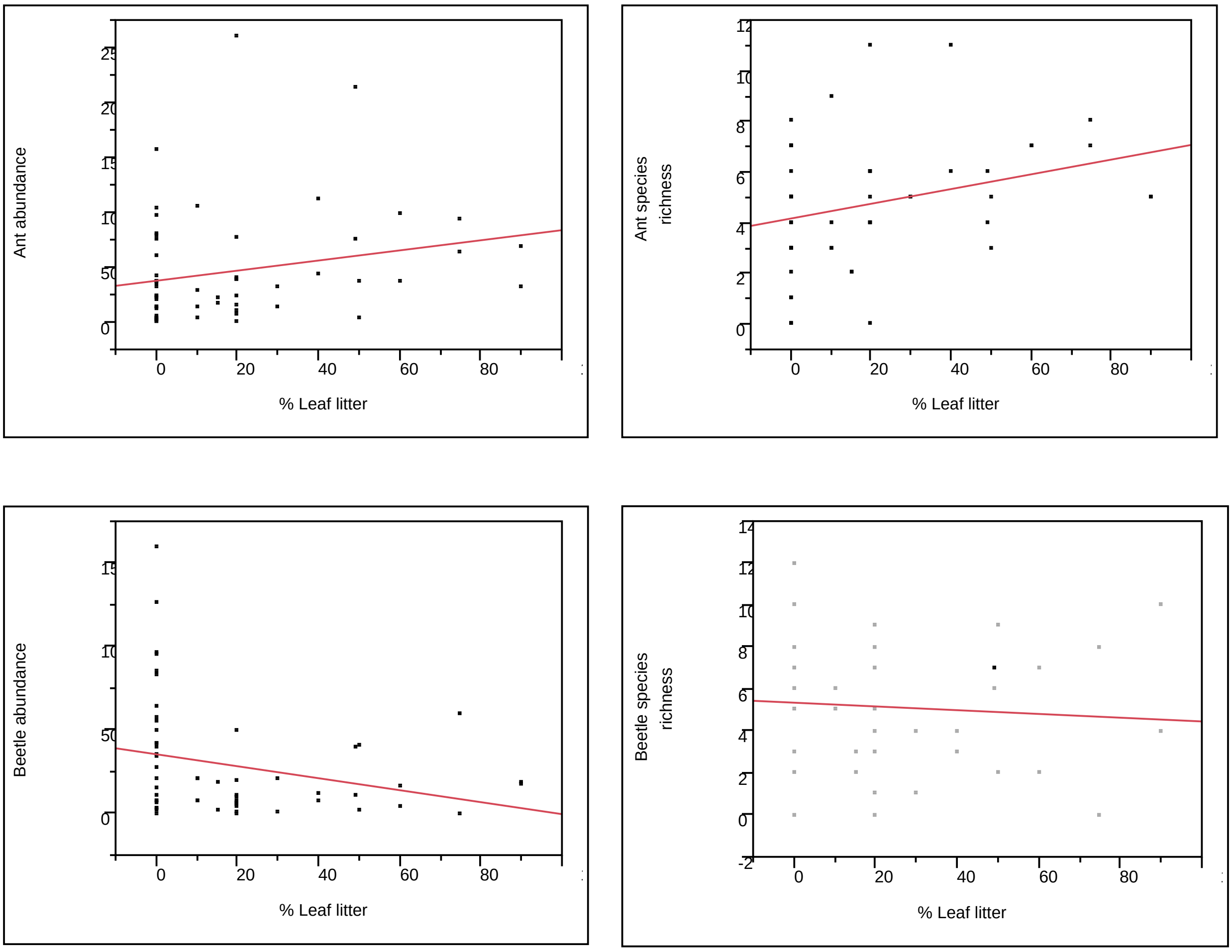
Regression showing relationship between planting age and ant and beetle abundance and species richness. Analysis conducted on data from planting sites only. Age is positively correlated with an abundance ($F_{1,54}$=5.09, p=0.0367), and negatively correlated with beetle abundance ($F_{1,54}$=6.39, p=0.0241) and beetle species richness ($F_{1,54}$=4.58, p=0.0460).

Appendix 10 cont.



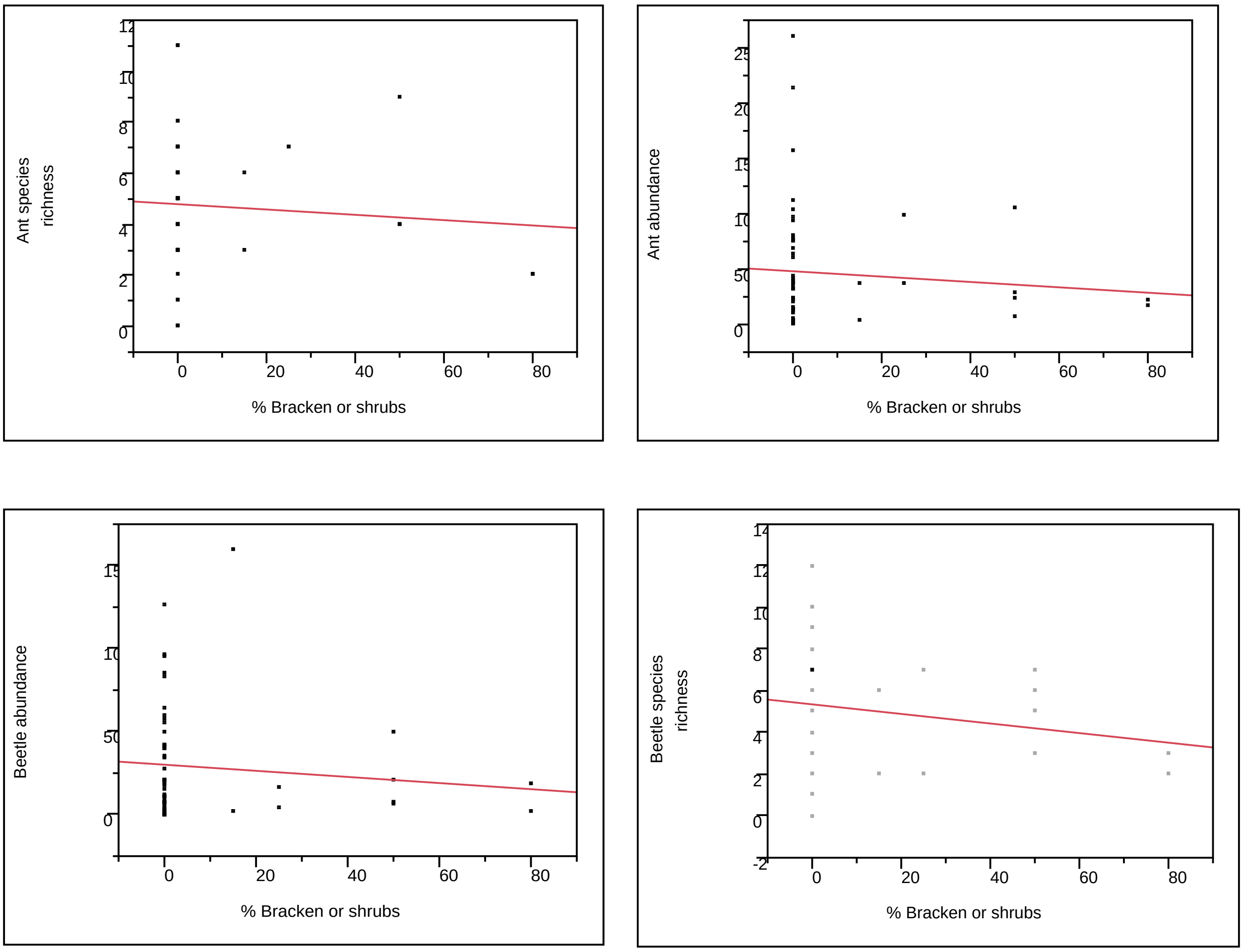
Regression showing relationship between ground cover (% grasses) and ant and beetle abundance and species richness. Analysis conducted on data from all sites. None of the relationships are significant.

Appendix 10 cont.



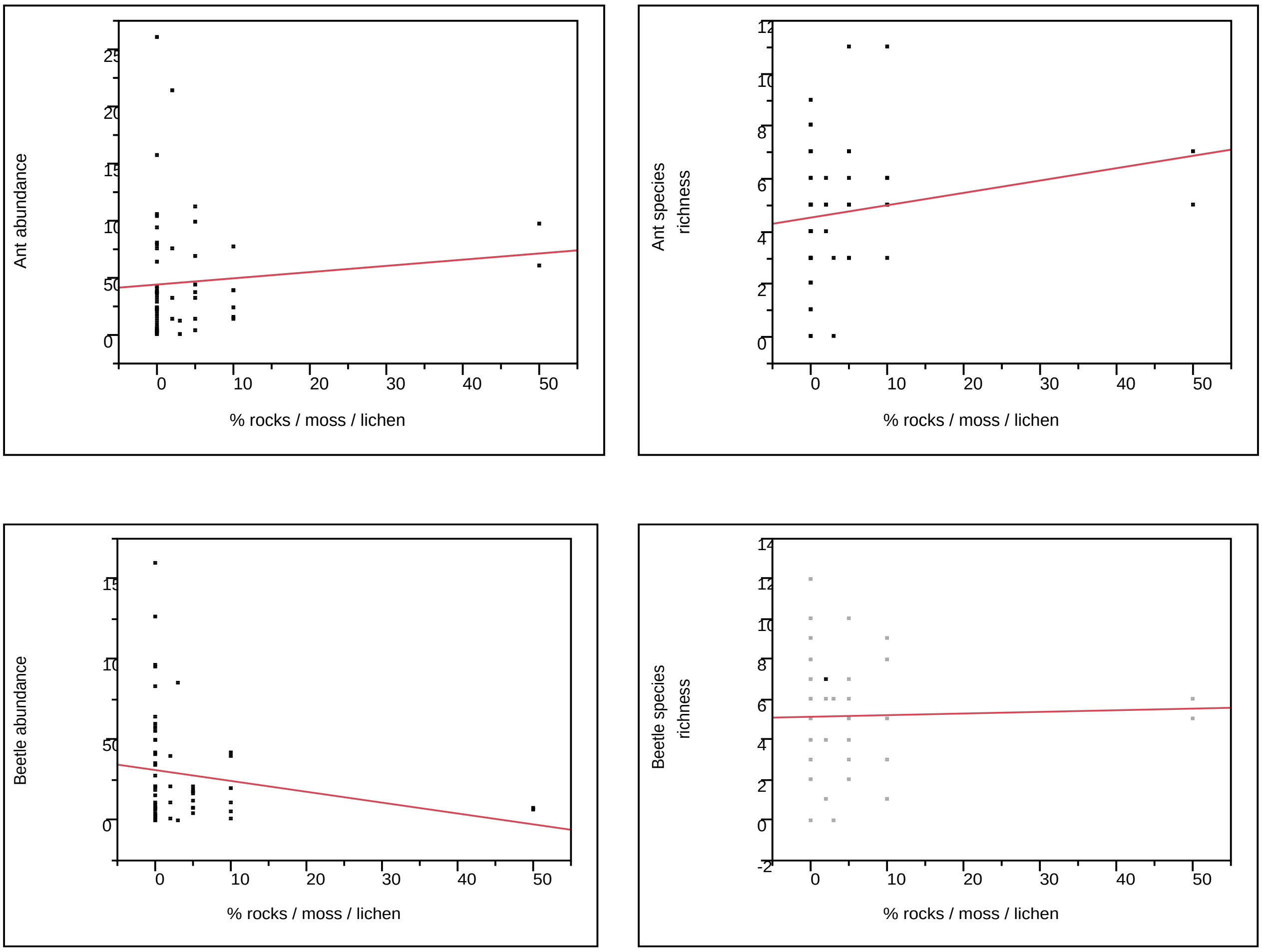
Regression showing relationship between ground cover (% leaf litter) and ant and beetle abundance and species richness. Analysis conducted on data from all sites. Leaf litter is positively correlated with ant species richness ($F_{1,54}$=5.50, p=0.0227) and negatively correlated with beetle abundance ($F_{1,54}$=4.08, p=0.04383) .

Appendix 10 cont.



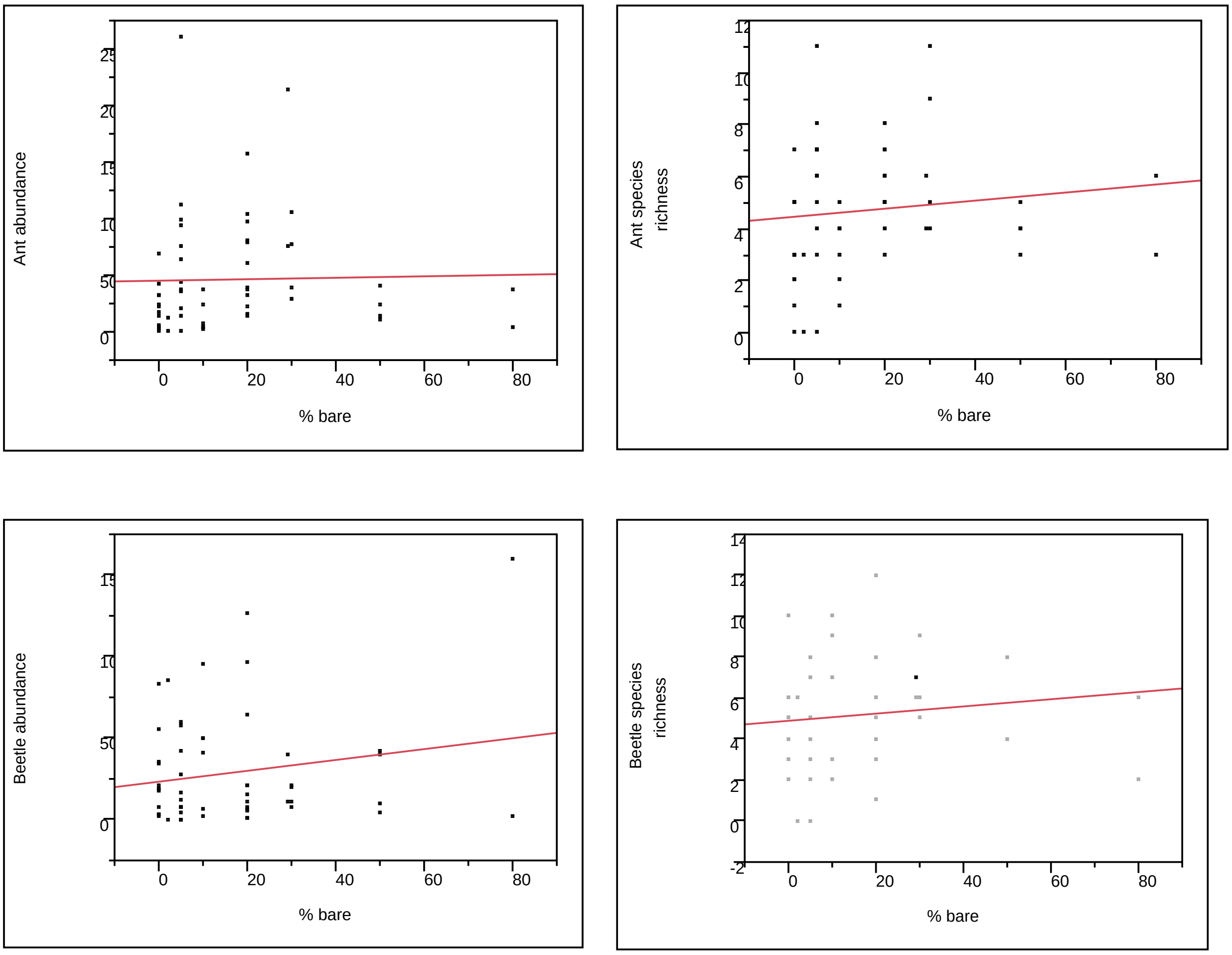
Regression showing relationship between ground cover (% bracken or shrubs) and ant and beetle abundance and species richness. Analysis conducted on data from all sites. None of the relationships are significant.

Appendix 10 cont.



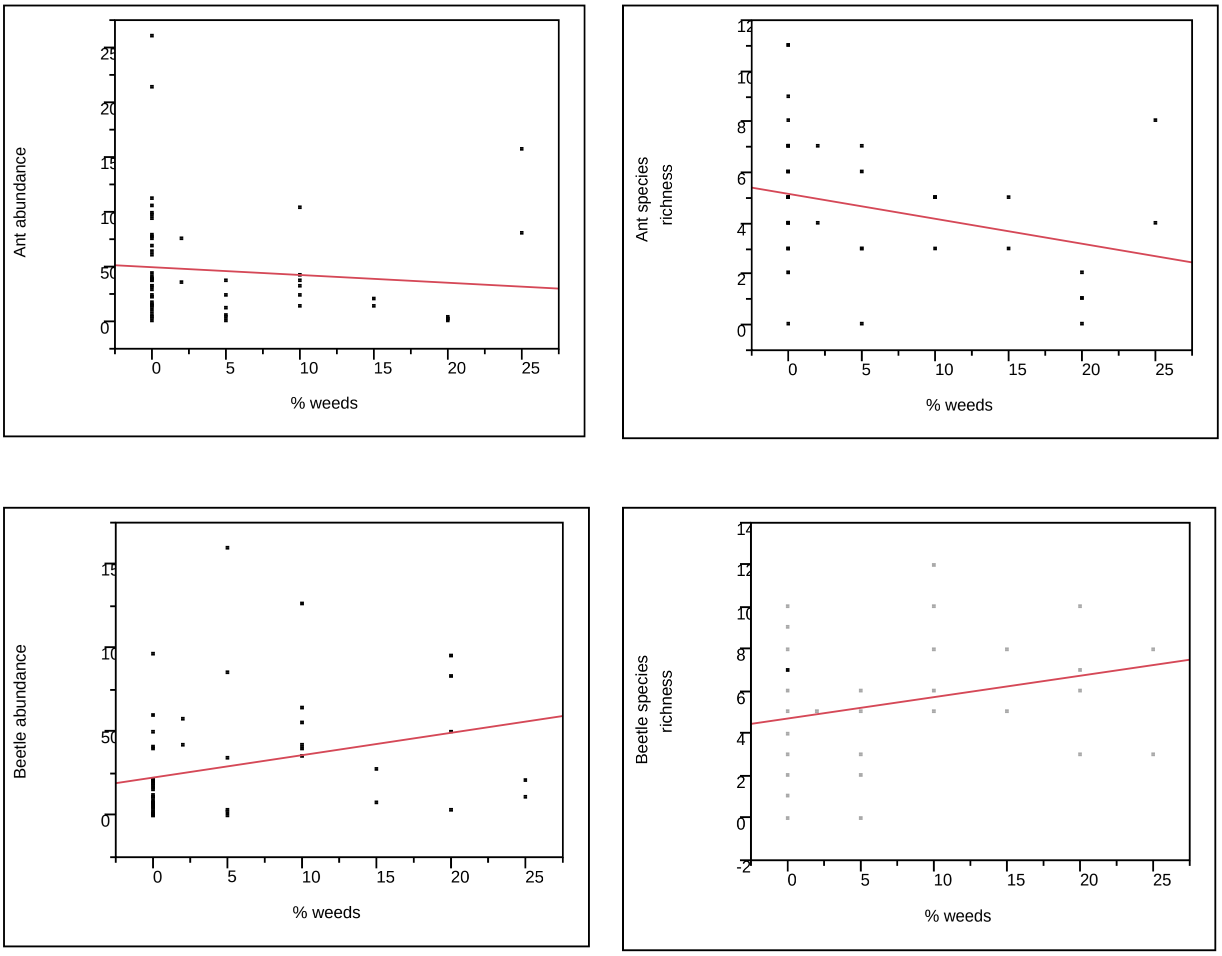
Regression showing relationship between ground cover (% rocks/moss/lichen) and ant and beetle abundance and species richness. Analysis conducted on data from all sites. None of the relationships are significant.

Appendix 10 cont.



Regression showing relationship between ground cover (% bare ground) and ant and beetle abundance and species richness. Analysis conducted on data from all sites. None of the relationships are significant.

Appendix 10 cont.

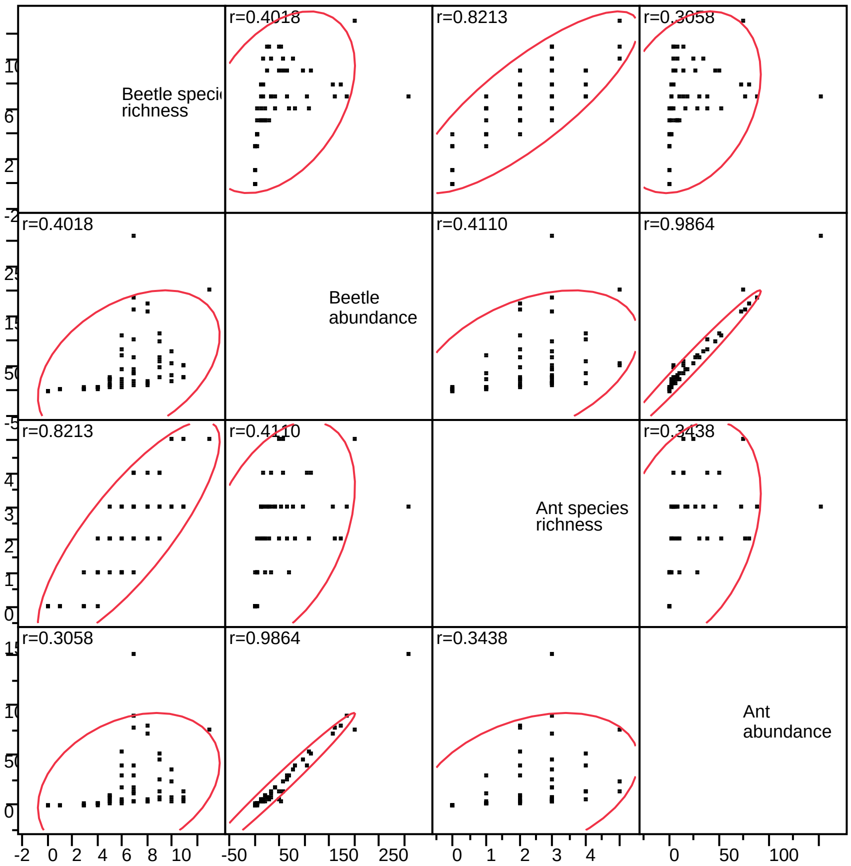


Regression showing relationship between ground cover (% weeds) and ant and beetle abundance and species richness. Analysis conducted on data from all sites. Weediness is negatively correlated with ant species richness ($F_{1,54}$=5.30, p=0.0251) and positively correlated with beetle abundance ($F_{1,54}$=4.87, p=0.0316) and beetle species richness ($F_{1,54}$=4.22, p=0.0448).

Appendix 11

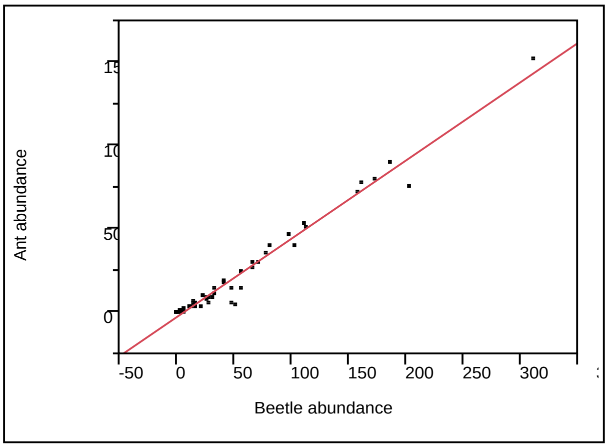
Relationship between ant abundance, beetle abundance, ant species richness and beetle species richness shown through multivariate correlations and regression analysis.

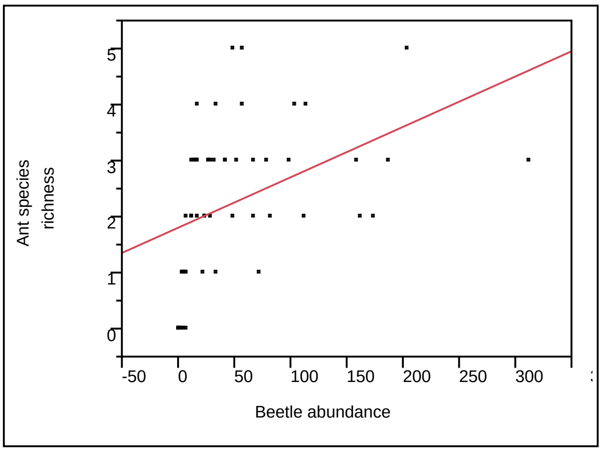
	Beetle species richness	Beetle abundance	Ant species richness	Ant abundance
Beetle species richness	1.0000			
Beetle abundance	0.4018	1.0000		
Ant species richness	0.8213	0.4110	1.0000	
Ant abundance	0.3058	0.9864	0.3438	1.0000

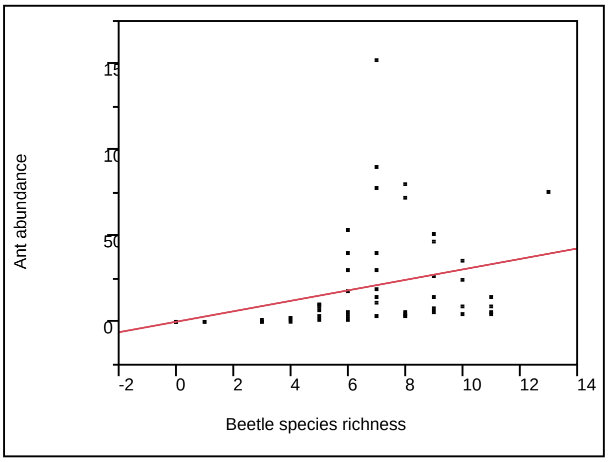


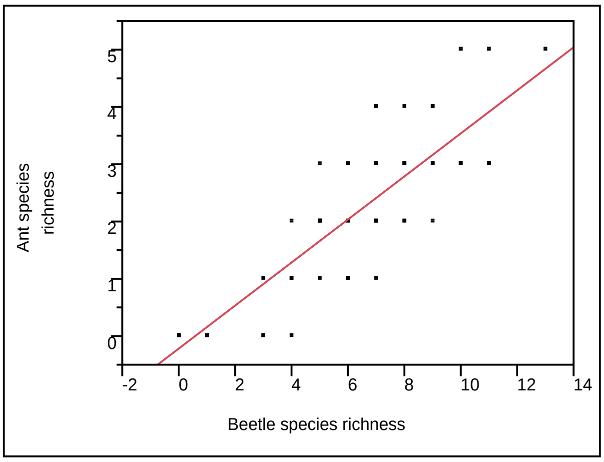
Appendix 11 cont.

	DF	R^2	F Ratio	Prob > F	Sig.
Ant abundance v Beetle abundance	1	0.9730	1944.949	<.0001	***
Ant abundance v Beetle species richness	1	0.0935	5.5705	0.0219	*
Beetle abundance v Beetle species richness	1	0.1690	10.9785	0.0016	**
Beetle abundance v Ant species richness	1	69.13429	111.8884	<.0001	***









Appendix 12

Catalogue of voucher specimens stored at UTAS Biogeography laboratory.

Catalogue Number	Species Name	Latitude	Longitude	Site	Elevation
801197	*Rhytidoponera victoriae*	42.638900°S	146.821429°E	site 1A	295m
801198	*Iridomyrmex bicknelli*	42.638900°S	146.821429°E	site 1A	295m
801199	*Rhytidoponera tasmaniensis*	42.638900°S	146.821429°E	site 1A	295m
801200	*Anonychomyrma biconvexa*	42.638900°S	146.821429°E	site 1A	295m
801201	*Pheidole*	42.638900°S	146.821429°E	site 1A	295m
801202	*Rhytidoponera victoriae*	42.638900°S	146.821429°E	site 1A	295m
801203	*Rhytidoponera tasmaniensis*	42.639465°S	146.819793°E	site 1B	300m
801204	*Iridomyrmex bicknelli*	42.639465°S	146.819793°E	site 1B	300m
801205	*Pheidole*	42.639465°S	146.819793°E	site 1B	300m
801206	*Rhytidoponera victoriae*	42.639465°S	146.819793°E	site 1B	300m
801207	*Anonychomyrma biconvexa*	42.639465°S	146.819793°E	site 1B	300m
801208	*Myrmecia forficata*	42.638219°S	146.818015°E	site 1C	315m
801209	*Amblyopone*	42.638219°S	146.818015°E	site 1C	315m
801210	*Prolasius nitidissimus*	42.638219°S	146.818015°E	site 1C	315m
801211	*Meranoplus*	42.638219°S	146.818015°E	site 1C	315m
801212	*Iridomyrmex bicknelli*	42.638219°S	146.818015°E	site 1C	315m
801213	*Pheidole*	42.638219°S	146.818015°E	site 1C	315m
801214	*Anonychomyrma* small	42.638219°S	146.818015°E	site 1C	315m
801215	*Iridomyrmex bicknelli*	42.638219°S	146.818015°E	site 1C	315m
801216	*Rhytidoponera victoriae*	42.540624°S	146.772326°E	site 2A	110m
801217	*Rhytidoponera tasmaniensis*	42.540624°S	146.772326°E	site 2A	110m
801218	*Iridomyrmex bicknelli*	42.540624°S	146.772326°E	site 2A	110m
801219	*Pheidole*	42.540624°S	146.772326°E	site 2A	110m
801220	*Tapinoma*	42.540624°S	146.772326°E	site 2A	110m
801221	*Tapinoma*	42.572415°S	146.772832°E	site 2C	100m
801222	*Anonychomyrma biconvexa*	42.572415°S	146.772832°E	site 2C	100m
801223	*Pheidole* orange	42.572415°S	146.772832°E	site 2C	100m
801224	*Rhytidoponera victoriae*	42.572415°S	146.772832°E	site 2C	100m
801225	*Rhytidoponera victoriae*	42.572415°S	146.772832°E	site 2C	100m
801226	*Myrmecia pilosula*	42.572415°S	146.772832°E	site 2C	100m
801227	*Pheidole*	42.572415°S	146.772832°E	site 2C	100m
801228	*Stigmacros* black	42.572415°S	146.772832°E	site 2C	100m
801229	*Stigmacros* brown	42.572415°S	146.772832°E	site 2C	100m
801230	*Tapinoma*	42.572415°S	146.772832°E	site 2C	100m
801231	*Rhytidoponera tasmaniensis*	42.563798°S	146.777980°E	site 10A	120m
801232	*Iridomyrmex bicknelli*	42.563798°S	146.777980°E	site 10A	120m
801233	*Pheidole*	42.563798°S	146.777980°E	site 10A	120m
801234	*Ochetellus*	42.563798°S	146.777980°E	site 10A	120m
801235	*Iridomyrmex vicina*	42.563798°S	146.777980°E	site 10A	120m
801236	*Camponotus consobrinus*	42.546163°S	146.849267°E	site 3A	120m
801237	*Myrmecia pilosula*	42.546163°S	146.849267°E	site 3A	120m
801238	*Rhytidoponera tasmaniensis*	42.546163°S	146.849267°E	site 3A	120m
801239	*Iridomyrmex bicknelli*	42.546163°S	146.849267°E	site 3A	120m

Appendix 12 cont.

Catalogue Number	Species Name	Latitude	Longitude	Site	Elevation
801240	*Pheidole*	42.546163°S	146.849267°E	site 3A	120m
801241	*Iridomyrmex vicina*	42.546163°S	146.849267°E	site 3A	120m
801242	*Rhytidoponera tasmaniensis*	42.546603°S	146.848316°E	site 3B	115m
801243	*Myrmecia pilosula*	42.546603°S	146.848316°E	site 3B	115m
801244	*Iridomyrmex dromus*	42.546603°S	146.848316°E	site 3B	115m
801245	*Notoncus ectatommoides*	42.546603°S	146.848316°E	site 3B	115m
801246	*Iridomyrmex bicknelli*	42.546603°S	146.848316°E	site 3B	115m
801247	*Pheidole*	42.546603°S	146.848316°E	site 3B	115m
801248	*Iridomyrmex vicina*	42.546603°S	146.848316°E	site 3B	115m
801249	*Melophorus*	42.545777°S	146.850462°E	site 3C	135m
801250	*Iridomyrmex bicknelli*	42.545777°S	146.850462°E	site 3C	135m
801251	*Pheidole*	42.545777°S	146.850462°E	site 3C	135m
801252	*Anonychomyrma biconvexa*	42.545777°S	146.850462°E	site 3C	135m
801253	*Myrmecia pilosula*	42.545777°S	146.850462°E	site 3C	135m
801254	*Rhytidoponera tasmaniensis*	42.545777°S	146.850462°E	site 3C	135m
801255	*Iridomyrmex vicina*	42.545777°S	146.850462°E	site 3C	135m
801256	*Monomorium* brown	42.545777°S	146.850462°E	site 3C	135m
801257	*Stigmacros* black	42.545777°S	146.850462°E	site 3C	135m
801258	*Stigmacros* brown	42.545777°S	146.850462°E	site 3C	135m
801259	*Myrmecia forficata*	42.271313°S	146.893676°E	site 4C	585m
801260	*Pheidole*	42.271313°S	146.893676°E	site 4C	585m
801261	*Stigmacros*	42.271313°S	146.893676°E	site 4C	585m
801262	*Stigmacros*	42.271313°S	146.893676°E	site 4C	585m
801263	*Anonychomyrma* small	42.271313°S	146.893676°E	site 4C	585m
801264	*Notoncus ectatommoides*	42.386892°S	147.017044°E	site 5A	360m
801265	*Camponotus elegans*	42.386892°S	147.017044°E	site 5A	360m
801266	*Pheidole*	42.386892°S	147.017044°E	site 5A	360m
801267	*Iridomyrmex bicknelli*	42.386892°S	147.017044°E	site 5A	360m
801268	*Iridomyrmex vicina*	42.386892°S	147.017044°E	site 5A	360m
801269	*Iridomyrmex bicknelli*	42.386892°S	147.017044°E	site 5A	360m
801270	*Rhytidoponera tasmaniensis*	42.402914°S	147.011532°E	site 5B	370m
801271	*Myrmecia pilosula*	42.402914°S	147.011532°E	site 5B	370m
801272	*Iridomyrmex bicknelli*	42.402914°S	147.011532°E	site 5B	370m
801273	*Rhytidoponera victoriae*	42.401995°S	147.021131°E	site 5C	440m
801274	*Rhytidoponera tasmaniensis*	42.401995°S	147.021131°E	site 5C	440m
801275	*Epobostruma*	42.401995°S	147.021131°E	site 5C	440m
801276	*Pheidole*	42.401995°S	147.021131°E	site 5C	440m
801277	*Anonychomyrma biconvexa*	42.401995°S	147.021131°E	site 5C	440m
801278	*Myrmecia pilosula*	42.496524°S	147.197260°E	site 6A	235m
801279	*Iridomyrmex bicknelli*	42.496524°S	147.197260°E	site 6A	235m
801280	*Pheidole*	42.496524°S	147.197260°E	site 6A	235m
801281	*Rhytidoponera victoriae*	42.496524°S	147.197260°E	site 6A	235m
801282	*Rhytidoponera tasmaniensis*	42.496524°S	147.197260°E	site 6A	235m
801283	*Rhytidoponera victoriae*	42.496524°S	147.197260°E	site 6A	235m
801284	*Rhytidoponera tasmaniensis*	42.496524°S	147.197260°E	site 6A	235m
801285	*Pheidole*	42.494907°S	147.200334°E	site 6B	240m
801286	*Rhytidoponera tasmaniensis*	42.494907°S	147.200334°E	site 6B	240m
801287	*Iridomyrmex vicina*	42.494907°S	147.200334°E	site 6B	240m

Appendix 12 cont.

Catalogue Number	Species Name	Latitude	Longitude	Site	Elevation
801288	*Iridomyrmex bicknelli*	42.494907°S	147.200334°E	site 6B	240m
801289	*Notoncus ectatommoides*	42.494907°S	147.200334°E	site 6B	240m
801290	*Polyrachis*	42.494907°S	147.200334°E	site 6B	240m
801291	*Myrmecia forficata*	42.388153°S	147.048340°E	site 6C	435m
801292	*Rhytidoponera victoriae*	42.388153°S	147.048340°E	site 6C	435m
801293	*Rhytidoponera tasmaniensis*	42.388153°S	147.048340°E	site 6C	435m
801294	*Monomorium* brown	42.388153°S	147.048340°E	site 6C	435m
801295	*Anonychomyrma biconvexa*	42.388153°S	147.048340°E	site 6C	435m
801296	*Iridomyrmex*	42.388153°S	147.048340°E	site 6C	435m
801297	*Rhytidoponera tasmaniensis*	42.398501°S	147.080305°E	site 7A	440m
801298	*Iridomyrmex bicknelli*	42.398501°S	147.080305°E	site 7A	440m
801299	*Pheidole*	42.398501°S	147.080305°E	site 7A	440m
801300	*Notoncus ectatommoides*	42.398501°S	147.080305°E	site 7A	440m
801301	*Iridomyrmex vicina*	42.398501°S	147.080305°E	site 7A	440m
801302	*Iridomyrmex*	42.398501°S	147.080305°E	site 7A	440m
801303	*Pheidole*	42.398501°S	147.080305°E	site 7A	440m
801304	*Rhytidoponera tasmaniensis*	42.398501°S	147.080305°E	site 7A	440m
801305	*Notoncus ectatommoides*	42.398501°S	147.080305°E	site 7A	440m
801306	*Iridomyrmex*	42.398501°S	147.080305°E	site 7A	440m
801307	*Iridomyrmex bicknelli*	42.398132°S	147.079551°E	site 7B	435m
801308	*Pheidole*	42.398132°S	147.079551°E	site 7B	435m
801309	*Monomorium* brown	42.398132°S	147.079551°E	site 7B	435m
801310	*Rhytidoponera tasmaniensis*	42.398132°S	147.079551°E	site 7B	435m
801311	*Iridomyrmex vicina*	42.398132°S	147.079551°E	site 7B	435m
801312	*Monomorium* brown	42.388274°S	147.037212°E	site 7C	430m
801313	*Rhytidoponera tasmaniensis*	42.388274°S	147.037212°E	site 7C	430m
801314	*Rhytidoponera victoriae*	42.388274°S	147.037212°E	site 7C	430m
801315	*Anonychomyrma biconvexa*	42.388274°S	147.037212°E	site 7C	430m
801316	*Myrmecia forficata*	42.242180°S	147.459499°E	site 8A	370m
801317	*Myrmecia pilosula*	42.242180°S	147.459499°E	site 8A	370m
801318	*Pheidole*	42.242180°S	147.459499°E	site 8A	370m
801319	*Iridomyrmex dromus*	42.242180°S	147.459499°E	site 8A	370m
801320	*Anonychomyrma biconvexa*	42.287971°S	147.403684°E	site 8B	500m
801321	*Prolasius* yellow	42.286666°S	147.403348°E	site 8C	510m
801322	*Myrmecia pilosula*	42.286666°S	147.403348°E	site 8C	510m
801323	*Pheidole*	42.279509°S	147.613903°E	site 9A	390m
801324	*Iridomyrmex bicknelli*	42.279509°S	147.613903°E	site 9A	390m
801325	*Iridomyrmex vicina*	42.279509°S	147.613903°E	site 9A	390m
801326	*Iridomyrmex dromus*	42.279509°S	147.613903°E	site 9A	390m
801327	*Myrmecia pilosula*	42.293492°S	147.406206°E	site 9B	490m
801328	*Rhytidoponera victoriae*	42.293492°S	147.406206°E	site 9B	490m
801329	*Pheidole*	42.293492°S	147.406206°E	site 9B	490m
801330	*Iridomyrmex bicknelli*	42.293492°S	147.406206°E	site 9B	490m
801331	*Iridomyrmex vicina*	42.293492°S	147.406206°E	site 9B	490m
801332	*Melophorus*	42.293492°S	147.406206°E	site 9B	490m
801333	*Anonychomyrma biconvexa*	42.293444°S	147.406970°E	site 9C	505m
801334	*Rhytidoponera victoriae*	42.293444°S	147.406970°E	site 9C	505m
801335	*Prolasius* yellow	42.293444°S	147.406970°E	site 9C	505m

Appendix 12 cont.

Catalogue Number	Species Name	Latitude	Longitude	Site	Elevation
801336	*Myrmecia pilosula*	42.268776°S	146.887169°E	site 4A	550m
801337	*Pheidole*	42.268776°S	146.887169°E	site 4A	550m
801338	*Iridomyrmex bicknelli*	42.268776°S	146.887169°E	site 4A	550m
801339	*Iridomyrmex bicknelli*	42.563032°S	146.778165°E	site 2B	130m
801340	*Iridomyrmex vicina*	42.563032°S	146.778165°E	site 2B	130m
801341	*Iridomyrmex dromus*	42.563032°S	146.778165°E	site 2B	130m
801342	*Pheidole*	42.563032°S	146.778165°E	site 2B	130m
801343	*Solenopsis*	42.563032°S	146.778165°E	site 2B	130m
801344	*Camponotus elegans*	42.572415°S	146.772832°E	site 2C	100m
801345	*Rhytidoponera tasmaniensis*	42.572415°S	146.772832°E	site 2C	100m
801346	*Monomorium* brown	42.572415°S	146.772832°E	site 2C	100m
801347	*Pheidole*	42.572415°S	146.772832°E	site 2C	100m
801348	*Iridomyrmex bicknelli*	42.563798°S	146.777980°E	site 2E	120m
801349	*Iridomyrmex vicina*	42.563798°S	146.777980°E	site 2E	120m
801350	*Pheidole*	42.563798°S	146.777980°E	site 2E	120m
801351	*Pheidole tasmaniensis*	42.638900°S	146.821429°E	site 1A	295m
801352	*Notoncus ectatommoides*	42.638900°S	146.821429°E	site 1A	295m
801353	*Rhytidoponera victoriae*	42.638900°S	146.821429°E	site 1A	295m
801354	*Rhytidoponera tasmaniensis*	42.639465°S	146.819793°E	site 1B	300m
801355	*Rhytidoponera victoriae*	42.639465°S	146.819793°E	site 1B	300m
801356	*Pheidole tasmaniensis*	42.639465°S	146.819793°E	site 1B	300m
801357	*Amblyopone*	42.638219°S	146.818015°E	site 1C	315m
801358	*Monomorium* brown	42.638219°S	146.818015°E	site 1C	315m
801359	*Anonychomyrma biconvexa*	42.638219°S	146.818015°E	site 1C	315m
801360	*Rhytidoponera victoriae*	42.638219°S	146.818015°E	site 1C	315m
801361	*Pheidole* large dark	42.5630°S	146.7782°E	site 2B	126m
801362	*Solenopsis*	42.5630°S	146.7782°E	site 2B	126m
801363	*Monomorium* brown large	42.5630°S	146.7782°E	site 2B	126m
801364	*Pheidole tasmaniensis*	42.563798°S	146.777980°E	site 2E	120m
801365	*Iridomyrmex vicina*	42.563798°S	146.777980°E	site 2E	120m
801366	*Iridomyrmex dromus*	42.563798°S	146.777980°E	site 2E	120m
801367	*Monomorium* brown large	42.563798°S	146.777980°E	site 2E	120m
801368	*Pheidole*	42.546163°S	146.849267°E	site 3A	120m
801369	*Tapinoma*	42.546163°S	146.849267°E	site 3A	120m
801370	*Camponotus consobrinus*	42.546163°S	146.849267°E	site 3A	120m
801371	*Rhytidoponera tasmaniensis*	42.546163°S	146.849267°E	site 3A	120m
801372	*Myrmecia pilosula*	42.546163°S	146.849267°E	site 3A	120m
801373	*Polyrachis*	42.546163°S	146.849267°E	site 3A	120m
801374	*Notoncus ectatommoides*	42.546603°S	146.848316°E	site 3B	115m
801375	*Iridomyrmex dromus*	42.546603°S	146.848316°E	site 3B	115m
801376	*Pheidole tasmaniensis*	42.546603°S	146.848316°E	site 3B	115m
801377	*Rhytidoponera tasmaniensis*	42.546603°S	146.848316°E	site 3B	115m
801378	*Pheidole tasmaniensis*	42.540624°S	146.772326°E	site 2A	110m
801379	*Iridomyrmex bicknelli*	42.540624°S	146.772326°E	site 2A	110m
801380	*Rhytidoponera victoriae*	42.540624°S	146.772326°E	site 2A	110m
801381	*Myrmecia pilosula*	42.540624°S	146.772326°E	site 2A	110m
801382	*Tapinoma*	42.540624°S	146.772326°E	site 2A	110m
801383	*Monomorium* brown	42.545777°S	146.850462°E	site 3C	135m

Appendix 12 cont.

Catalogue Number	Species Name	Latitude	Longitude	Site	Elevation
801384	*Camponotus elegans*	42.545777°S	146.850462°E	site 3C	135m
801385	*Camponotus consobrinus*	42.545777°S	146.850462°E	site 3C	135m
801386	*Pheidole tasmaniensis*	42.545777°S	146.850462°E	site 3C	135m
801387	*Rhytidoponera tasmaniensis*	42.545777°S	146.850462°E	site 3C	135m
801388	*Myrmecia pilosula*	42.268776°S	146.887169°E	site 4A	550m
801389	*Notoncus ectatommoides*	42.268776°S	146.887169°E	site 4A	550m
801390	*Pheidole tasmaniensis*	42.268776°S	146.887169°E	site 4A	550m
801391	*Iridomyrmex vicina*	42.268776°S	146.887169°E	site 4A	550m
801392	*Rhytidoponera victoriae*	42.268776°S	146.887169°E	site 4A	550m
801393	*Monomorium* brown large	42.26900°S	146.88630°E	site 4B	540m
801394	*Notoncus ectatommoides*	42.26900°S	146.88630°E	site 4B	540m
801395	*Anonychomyrma* small	42.271313°S	146.893676°E	site 4C	585m
801396	*Notoncus ectatommoides*	42.271313°S	146.893676°E	site 4C	585m
801397	*Iridomyrmex vicina*	42.271313°S	146.893676°E	site 4C	585m
801398	*Pheidole tasmaniensis*	42.271313°S	146.893676°E	site 4C	585m
801399	*Notoncus ectatommoides*	42.386892°S	147.017044°E	site 5A	360m
801400	*Camponotus elegans*	42.386892°S	147.017044°E	site 5A	360m
801401	*Monomorium* brown large	42.386892°S	147.017044°E	site 5A	360m
801402	*Rhytidoponera tasmaniensis*	42.401995°S	147.021131°E	site 5C	440m
801403	*Notoncus ectatommoides*	42.401995°S	147.021131°E	site 5C	440m
801404	*Monomorium* brown	42.401995°S	147.021131°E	site 5C	440m
801405	*Anonychomyrma biconvexa*	42.401995°S	147.021131°E	site 5C	440m
801406	*Pheidole* large dark	42.402914°S	147.011532°E	site 5B	370m
801407	*Iridomyrmex vicina*	42.402914°S	147.011532°E	site 5B	370m
801408	*Iridomyrmex vicina*	42.386892°S	147.017044°E	site 5A	360m
801409	*Iridomyrmex dromus*	42.386892°S	147.017044°E	site 5A	360m
801410	*Iridomyrmex bicknelli*	42.386892°S	147.017044°E	site 5A	360m
801411	*Solenopsis*	42.386892°S	147.017044°E	site 5A	360m
801412	*Pheidole* large dark	42.386892°S	147.017044°E	site 5A	360m
801413	*Notoncus ectatommoides*	42.494907°S	147.200334°E	site 6B	240m
801414	*Rhytidoponera tasmaniensis*	42.494907°S	147.200334°E	site 6B	240m
801415	*Pheidole tasmaniensis*	42.494907°S	147.200334°E	site 6B	240m
801416	*Myrmecia pilosula*	42.494907°S	147.200334°E	site 6B	240m
801417	*Rhytidoponera victoriae*	42.388153°S	147.048340°E	site 6C	435m
801418	*Rhytidoponera tasmaniensis*	42.388153°S	147.048340°E	site 6C	435m
801419	*Anonychomyrma biconvexa*	42.388153°S	147.048340°E	site 6C	435m
801420	*Monomorium* brown	42.388153°S	147.048340°E	site 6C	435m
801421	*Myrmecia forficata*	42.388153°S	147.048340°E	site 6C	435m
801422	*Cerapachys*	42.388153°S	147.048340°E	site 6C	435m
801423	*Rhytidoponera tasmaniensis*	42.388153°S	147.048340°E	site 6C	435m
801424	*Iridomyrmex vicina*	42.388153°S	147.048340°E	site 6C	435m
801425	*Rhytidoponera victoriae*	42.388153°S	147.048340°E	site 6C	435m
801426	*Iridomyrmex vicina*	42.398501°S	147.080305°E	site 7A	440m
801427	*Notoncus ectatommoides*	42.398501°S	147.080305°E	site 7A	440m
801428	*Rhytidoponera tasmaniensis*	42.398501°S	147.080305°E	site 7A	440m
801429	*Pheidole tasmaniensis*	42.398501°S	147.080305°E	site 7A	440m
801430	*Monomorium* brown large	42.398501°S	147.080305°E	site 7A	440m
801431	*Rhytidoponera tasmaniensis*	42.398132°S	147.079551°E	site 7B	435m

Appendix 12 cont.

Catalogue Number	Species Name	Latitude	Longitude	Site	Elevation
801432	*Pheidole*	42.398132°S	147.079551°E	site 7B	435m
801433	*Monomorium* brown large	42.398132°S	147.079551°E	site 7B	435m
801434	*Notoncus ectatommoides*	42.398132°S	147.079551°E	site 7B	435m
801435	*Camponotus elegans*	42.388274°S	147.037212°E	site 7C	430m
801436	*Rhytidoponera victoriae*	42.388274°S	147.037212°E	site 7C	430m
801437	*Iridomyrmex bicknelli*	42.388274°S	147.037212°E	site 7C	430m
801438	*Myrmecia pilosula*	42.388274°S	147.037212°E	site 7C	430m
801439	*Rhytidoponera tasmaniensis*	42.388274°S	147.037212°E	site 7C	430m
801440	*Anonychomyrma biconvexa*	42.388274°S	147.037212°E	site 7C	430m
801441	*Tapinoma*	42.388274°S	147.037212°E	site 7C	430m
801442	*Pheidole tasmaniensis*	42.242180°S	147.459499°E	site 8A	370m
801443	*Iridomyrmex dromus*	42.242180°S	147.459499°E	site 8A	370m
801444	*Pheidole* large dark	42.242180°S	147.459499°E	site 8A	370m
801445	*Myrmecia pilosula*	42.242180°S	147.459499°E	site 8A	370m
801446	*Amblyopone*	42.279509°S	147.613903°E	site 9A	390m
801447	*Pheidole*	42.293492°S	147.406206°E	site 9B	490m
801448	*Notoncus ectatommoides*	42.293492°S	147.406206°E	site 9B	490m
801449	*Rhytidoponera victoriae*	42.293492°S	147.406206°E	site 9B	490m